Spring 2020

MS Environmental Biology Capstone Project

Chase Westbrook
Regis University

MS ENVIRONMENTAL BIOLOGY
CAPSTONE PROJECT

by

Chase W. Westbrook

A Project Presented in Partial Fulfillment
of the Requirements for the Degree
Masters of Science
in Environmental Biology

REGIS UNIVERSITY
May, 2020

Table of Contents

CHAPTER 2, LIST OF TABLES

CHAPTER 2, LIST OF FIGURES

CHAPTER 3, LIST OF FIGURES

CHAPTER 4, LIST OF FIGURES

CHAPTER 1. LITERATURE REVIEW

Human Land Use & Pollution Impacts on Freshwater Vernal Pools in the United States

Today more than half of the world's original wetlands no longer exist. Due to a variety of natural stressors including processes such as erosion, droughts, storms, and changes in sea level, wetland extent has been dramatically affected (Keddy, 2010). However, the vast majority of wetland losses over the past several centuries have occurred as a result of anthropogenic activity (Keddy, 2010). Vernal pool systems, ephemeral freshwater wetlands that form seasonally in topographic depressions (Kneitel & Lessin, 2009), have become particularly threatened by human activities because of the increased land conversion rates to agriculture and/or urban land on or in close proximity to vernal pool habitat. Vernal pools are found all over the world and primarily occur in Mediterranean climate conditions, but can also be found in previously glaciated areas of northeastern and midwestern regions of the U.S. (Petrick, Silveira & USFWS, 1998). There is a strong call from academic and federal entities for the protection and conservation of these systems, because of the important roles and services they provide. The most significant threats to vernal pools are habitat destruction from land cover change and degrading land conditions from pollution (Duffy & Kahara, 2011). While many studies examine human influence on wetland ecosystems, there have been few studies on anthropogenic activities and their impacts on freshwater vernal pools in the United States. This review will explain how humans impact these empirically underrepresented systems in the U.S., and will argue that freshwater vernal pools need to be analyzed independently from other wetland types to properly assess the problems these distinct wetland types face.

Vernal Pools: Unique Wetlands with Unique Functions

Two principal types of wetlands exist: marine/estuarine (saltwater) and freshwater. Marine and estuarine wetland habitats account for a mere 5% of the total wetland coverage of the contiguous United States, while freshwater wetlands cover the 95% majority (Dahl, 2011). Vernal pools can persist in both marine and freshwater environments but are more common within the latter. Freshwater vernal pools occur all over the world, but because of the specific climatic conditions that these pools require, they are limited primarily to the west coast, and previously glaciated areas of the northeastern and midwestern U.S. where depressions formed by glacial retreat occur (Kneitel & Lessin, 2009). Freshwater vernal pools are precipitation-filled seasonal wetlands inundated during the aquatic phase when temperature is sufficient for plant growth, followed by a brief waterlogged-terrestrial stage and then leading up to extreme desiccating soil conditions of extended duration (Keeley & Zedler 1998). They range in size from small puddles to shallow lakes and are usually found within grassland ecosystems (Duffy & Kahara, 2011). These attributes create harsh conditions for species development, but vernal pools still provide all ecological services typical of wetlands (Bauder, 2005), making these unique systems vitally important to protect.

Wetlands provide a multitude of important ecological functions, including providing habitat for a wide variety of flora and fauna; filtering, cleaning and storing water; collecting and holding flood waters; sequestering atmospheric carbon; and providing places of beauty and recreation (Cronk & Mitsch, 1992). Vernal pools have the potential to provide all of these services, but what makes them unique are the wildlife species these habitats harbor. Vernal pools are biodiversity hotspots that provide suitable habitat for a variety of endemic and rare species (Schierenbeck, 2017). Vernal pool communities include both opportunistic species (e.g. flying insects, birds, amphibians) which are present during the aquatic phase and leave as the pool dries, and less-vagile species (e.g. plants, crustaceans, gastropods), which survive the drought phase as dormant adults, juveniles, or propagules (King, Simovich & Brusca,1996). They serve as habitat for many amphibian and macroinvertebrate species, because vernal pools are devoid of fish that may prey on natal organisms (Cronk & Mitsch, 1992). In California, vernal pools provide habitat to multiple endemic, threatened macroinvertebrate species that are distributed sporadically and are relatively rare, such as *Branchinecta lynchi*, *Lepidurus packardi*, *Cyzicus californicus* and *Linderiella occidentalis* (Anostraca) (King, 1998). *Linderiella occidentalis*, commonly known as the California fairy shrimp, is one of the primary (bio)indicators for the longevity of vernal pool inundation, because it is adapted to reach maturity and reproduce faster than other vernal pool branchiopods, spending the majority of its lifecycle within a single pool (King et al., 1996); this could be useful information when looking at vernal pool health. Species which are widespread, common, and occur in a large fraction of habitat patches are less likely to be threatened with extinction due to habitat loss than are species which are endemic, rare, and occur in only a small fraction of patches, such as species that are found in vernal pool habitats (King, 1998). The presence of endemic and threatened species that have unique lifecycles associated with these systems is one of the major reasons why it is important to analyze vernal pools as independently distinct wetland systems.

History of Wetland Loss

Following European settlement, an extremely significant portion of wetland systems were destroyed by humans in the U.S. The extent of this devastation is alarming; nearly 87% of the entire planet’s original wetland habitats have been eradicated in the past 300 years, with approximately 54% of the wetlands being lost in the last 30 years (Hu et.al., 2017). In recent decades, humans have drained and/or filled wetlands to provide land for the exponentially growing human population, who utilized it for agriculture, residential development, mining, and dumping (King, 1998). In the 1970's, strong pleas from citizen groups forced legislators to realize the importance of protecting these valuable habitats, which prompted a variety of federal and state Wetland Protection Legislation to be enacted (Dahl & Allord, 1996). Despite these laws, wetlands are still being negatively affected. Although direct wetland loss due to wetland filling has decreased, other human activities that indirectly affect wetlands are equally dangerous. Indirect impacts such as agricultural and residential runoff, fossil fuel combustion and hydrological pattern alterations all play a role in the degradation and eventual demise of these wetland habitats (King, 1998). Large wetland complexes are typically state and/or federally protected, while smaller patches of vernal pool habitat tend to be overlooked (Dahl & Allord, 1996). For example, agricultural runoff can readily drain into vernal pool habitats because a large number of vernal pools are found near agricultural land (King, 1998). Vernal pool habitats' close contact with agricultural land has put them in this detrimental situation.

Vernal pool habitats are commonly found in close proximity to developed land due to their small, habitually isolated disposition causing them to go unnoticed and enabling land developers to take advantage. A recent assessment of the status of freshwater vernal pools in the U.S. based on vernal pool population in California estimates that more than 90% of vernal pool habitat in the Central Valley of California and in other parts of the state have been lost (CDFW, 2013). In 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was established, outlining habitat protection techniques and a variety of endemic and threatened species within these habitats (USFWS, 2017). California legislators may understand the threat to vernal pools and their associated inhabitants; however, these habitats still fall victim to indirect anthropogenic impacts. The majority of developed land where vernal pool habitats are commonly found has been privatized, which does not allow legislative powers to monitor vernal pool status (USFWS, 2017). Vernal pools are typically small, which makes placing federal jurisdiction over them nearly impossible, compared to large palustrine systems that are readily qualified to obtain that designation (Johnson, 1993). Federal jurisdiction designation significantly increases the protection for wetlands, providing resources such as one hundred-foot buffer zone around the jurisdictional delineation where no development may occur (Johnson, 1993). Even though vernal pool habitats are federally protected in California, local conditions influenced by land cover change surrounding vernal pool habitats have been the cause of this continual degradation, placing them in this unprecedented situation.

Changing Land Cover Effects on Vernal Pools

In the U.S., agriculture and livestock production are the two most dominant drivers of land cover change that result in wetland habitat loss and degradation. Many farmlands in the Western U.S. contain or are in close proximity to freshwater wetland systems, including vernal pool habitat (Javornika & Collinge, 2016). Overgrazing by cattle can alter wetland species composition, which can lead to decreases in density, biomass, and species richness (Bauder, 2005). More than half (51%) of Western U.S. rangelands managed by the Bureau of Land Management and the Forest Service were in suboptimal conditions because of degrading land conditions (Fleischner, 1994). These conditions, such as high nutrient loads, increased erosion and altered hydrological patterns, are highly unsuitable for most species and are particularly pertinent to endemic species (Fleischner, 1994). Land cover change has distinct impacts on vernal pools compared to other wetlands because of the unique habitat conditions vernal pools provide inhabitants. Vernal pool wildlife requires specific nutrient levels and hydrology in order to thrive in these habitats and any alteration can prove fatal to organisms that require those conditions to survive (Kneitel & Lessin, 2009). Altered land conditions from changing land cover type can introduce invasive species, which has the potential to harm vernal pool habitat by threatening native and endemic biodiversity, and altering ecosystem functionality (Collinge, Ray, & Gerhardt, 2011). While using livestock to graze overgrown areas may prevent the introduction and spread of exotic species, overgrazing and other farmland activities can disrupt ecosystem functioning and structure, including interference in nutrient cycling and ecological succession (Fleischner, 1994). Disrupting vernal pool habitats within these landscapes commonly alters hydrology of the land surrounding vernal pools, increasing erosion and runoff, and contributing pesticides and fertilizers (Carpenter et al., 1998). The habitat requirements for vernal pool inhabitants are unique to these systems and the continued alteration of these specific habitat requirements shows that vernal pools are not yet widely understood. Altering land cover types affect vernal pool habitat in a distinct way and these changes also lead to additional impacts from polluted runoff, consequently effecting these habitats in a particular fashion.

Pollutant Effects on Vernal Pools

Pollutants are prevalent throughout the natural environment. Most studies of pollutants' effects on waterbodies use perennial systems, such as large lakes and rivers, but ephemeral systems are also affected by anthropogenic pollution (Kneitel & Lessin, 2009). For example, excessive influx of nutrient pollution into vernal pool ecosystems can lead to cultural eutrophication (Schindler, 2006). Fertilizers used for agriculture typically contain high concentrations of phosphorous and nitrate in order to stimulate plant growth (Savcı, 2012). When nutrients were added to soil-filled mesocosms designed to mimic vernal pools, higher concentrations of phosphate and nitrate decreased vascular plant species richness and percent of plant cover (Kneitel & Lessin, 2009) by favoring algal mat growth. Large algal blooms in these fragile systems can negatively affect them by impeding sunlight penetration and depleting oxygen levels, leading to the displacement of aquatic species (Dyble, Fulton, Moisander & Paerl, 2001). Algal blooms are common in most wetland systems and typically do not cause irreparable damage to the habitat, but excessive nutrient deposition resulting in large algal blooms in a vernal pool could prove to be devastating to these unique habitats (Johnson et al., 2005). Increased nutrient loads from polluted runoff have distinct effects on vernal pools' seasonal phases, where nutrient pollutants are able to carry over from phase to phase, affecting the subsequent phase (Kneitel & Lessin, 2009). Shifts in the terrestrial phase affect the aquatic phase and vice versa (Kneitel & Lessin, 2009. Nutrient loading has detrimental effects on vernal pool habitats, but other pollutants such as heavy metals can devastate these habitats as well.

Metals are introduced in aquatic systems as a result of mineral weathering, volcanic eruptions, and from a variety of human activities involving mining, processing, or the use of materials that contain metal pollutants. The most common metal pollutant found in the environment is mercury. Methylmercury (MeHg) is a toxic form of mercury that is formed in anaerobic conditions, which are common in the waterlogged sediments of wetlands (Benoit et al., 2013). Although MeHg is naturally produced in vernal pools, the combination of this natural production and mercury deposition from the atmosphere and human activity can create a toxic environment for the species associated with vernal pool habitats (Benoit et al., 2013). Wildlife exposed to high MeHg concentrations are at risk of bioaccumulation, where the MeHg can magnify effects up the food chain as larger organisms consume smaller ones (Brooks et al., 2012). As predatory animals eat other organisms, MeHg moves from organism to organism, increasing concentrations (Brooks et al., 2012). This can severely affect wildlife using the pool for sustenance, and while this has been extensively studied in avian species in larger wetland complexes, e.g. eggshell thinning in waterfowl (Ackerman et al., 2016), the effects of MeHg in vernal pool habitats have been widely underreported. Vernal pools are important hotspots of MeHg bioaccumulation, and biota from the pools may be vectors of MeHg to the terrestrial ecosystem. Endemic amphibians that spend the majority of their life cycle in vernal pools have high concentrations of MeHg they bioaccumulate, which can be detrimental to their fitness and to their predators (Faccio et al., 2019). This information also suggests that amphibians could potentially be important bioindicators for monitoring MeHg loading and bioavailability in these ecosystems (Faccio et al., 2019). This may prove to be important for land managers deciding on conservation plans, and also demonstrates that metal pollutants within vernal pool habitat needs further research in order to understand this complex relationship.

Conclusion

A famous quote from Rachel Carson's *Silent Spring* (1962) - "In nature, nothing exists alone." - perfectly exemplifies that any action subjected on the natural environment can cause a cascade of negative outcomes. This inference has a direct relationship to vernal pool habitats, where direct effects to the natural landscape can indirectly affect these systems. Vernal pool habitats require very specific climatic conditions in order to persist (Petrick, Silveira & USFWS, 1998). Providing a compilation of wetland types in a study may be helpful for the overall goal of wetland habitat protection and conservation, but a more independent, detailed synthesis of vernal pool systems is required in order to properly assess each facet of these systems. Future research must be conducted specifically on the effects on vernal pools, particularly research in land cover change and associated pollutants, to prevent the extinction of this critical wetland habitat.

References

Ackerman, J.T., Eagles-Smith, C.A., Herzog, M.P., Hartman, C.A., Peterson, S.H., Evers, D.C., et al. (2016). Avian mercury exposure and toxicological risk across western North America: A synthesis. *Science of The Total Environment*, *568*, 749-769.

Bauder, E.T. (2005). The effects of an unpredictable precipitation regime on vernal pool hydrology. *Freshwater Biology*, *50*(12), 2129-2135.

Benoit, J.M., Cato, D.A., Denison, K.C., & Moreira, A.E. (2013). Seasonal mercury dynamics in a New England vernal pool. *Wetlands*, *33*(5), 887-894.

Brooks, R.T., Eggert, S.L., Nislow, K.H., Kolka, R.K., Chen, C.Y. & Ward, D.M. (2012). Preliminary assessment of mercury accumulation in Massachusetts and Minnesota seasonal forest pools. *Wetlands*, *32*(4), 653-663.

California Department of Fish and Wildlife. (2013). California's Vernal Pools. Retrieved from https://www.wildlife.ca.gov/Conservation/Plants/Vernal-Pools (Original work published 2013).

Carpenter, S., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N. & Smith, V.H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications, 8*(3), 559-568.

Carson, R. (1962). *Silent Spring*. Boston: Houghton Mifflin Company.

Collinge, S.K., Ray, C. & Gerhardt, F. (2011). Long-term dynamics of biotic and abiotic resistance to exotic species invasion in restored vernal pool plant communities. *Ecological Applications, 21*(6), 2105-2118.

Cronk, J.K. & Mitsch, W.J. (1992). Creation and restoration of wetlands: some design consideration for ecological engineering. *Soil Restoration, 17*, 217-259.

Dahl, T. E. (2011). Status and trends of wetlands in the conterminous United States 2004 to 2009 [PDF file]. U.S. Department of the Interior, Fish and Wildlife Service. Retrieved from https://www.fws.gov/wetlands/Documents/Status-and-Trends-of-Wetlands-in-the-Conterminous-United-States-2004-to-2009.pdf. (Original work published 2011).

Dahl, T. E. & Allord, G. J. (1996). Technical aspects of wetlands history of wetlands in the conterminous United States. United States Geological Survey Water Supply, Paper 2425.

Duffy, W. G. & Kahara, S.N. (2011). Wetland ecosystem services in California's Central Valley and implications for the Wetland Reserve Program. *Ecological Applications, 21*(sp1), S128-S134.

Dyble, J., Fulton, R.S., Moisander, P.H. & Paerl, H.W. (2001). Harmful freshwater algal blooms, with an emphasis on cyanobacteria. *The Scientific World Journal, 1*, 76-113.

Faccio, S.D., Buckman, K.L., Lloyd, J.D., Curtis, A.N. & Taylor, V.F. (2019). Bioaccumulation of methylmercury in wood frogs and spotted salamanders in Vermont vernal pools. *Ecotoxicology, 28*(7), 717-731

Fleischner, T. (1994). Ecological costs of livestock grazing in western North America. *Conservation Biology, 8*(3), 629-644.

Hu, S., Niu, Z., Chen, Y., Li, L. & Zhang, H. (2017). Global wetlands: Potential distribution, wetland loss, and status. *Science of The Total Environment, 586*, 319-327.

Javornika, C.J. & Collinge, S.K. (2016). Influences of annual weather variability on vernal pool plant abundance and community composition. *Aquatic Botany, 134*, 61-67.

Johnson, S. (1993). Federal regulation of isolated wetlands. *Environmental Law, 23*(1), 1-42.

Johnson W.C., Millett B.V., Gilmanov T., Voldseth R.A., Guntenspergen G.R. & Naugle D.E.

(2005). Vulnerability of northern prairie wetlands to climate change. *BioScience, 55*(10), 863-872.

Keddy P.A. (2010). *Wetland Ecology: Principles and Conservation*. New York: Cambridge University Press.

Keeley, J.E., & P.H., Zedler (1998). Characterization and global distribution of vernal pools. 1-14. In Carol C. Witham (ed.) Vernal Pool Ecosystems: proceedings from a 1996 conference. California Native Plant Society, Sacramento, CA.

King, J.L. (1998). Loss of Diversity as a Consequence of Habitat Destruction in California Vernal Pools. *California Native Plant Society*. Proceedings from a 1996 Conference, Sacramento, CA.

King, J.L., Simovich, M.A., & Brusca, R.C. (1996). Species richness, endemism and ecology of crustacean assemblages in northern California vernal pools. *Hydrobiologia, 328*(2), 85-116.

Kneitel J.M. & Lessin C.L. (2009). Ecosystem-phase interactions: aquatic eutrophication decreases terrestrial plant diversity in California vernal pools. *Oecologia, 163*(2), 461-9.

Petrick, J.A., Silveira, J.G. & USFWS. (1998). Avian uses of vernal pools and implications for conservation practice. *California Native Plant Society*. Proceedings from a 1996 Conference, Sacramento, CA.

Savci, S. (2012). An agricultural pollutant: Chemical fertilizer. *International Journal of Environmental Science and Development. 3*(1), 77-80.

Schierenbeck, K.A. (2017). Population-level genetic variation and climate change in a biodiversity hotspot. *Annals of Botany, 119*(2), 215-228.

Schindler, D.W. (2006). Recent advances in the understanding and management of eutrophication. *Limnology and Oceanography*, *51*(1part2), 356-363.

U.S. Fish & Wildlife Service. (2017). Vernal Pool Recovery Plan. Retrieved from https://www.fws.gov/sacramento/es/Recovery-Planning/Vernal-Pool/ (Original work published 2005).

CHAPTER 2. GRANT PROPOSAL

Habitat Assessment for Kettle Pond Ecological Functionality in Boulder County Parks Open Space

Chase Westbrook

cwestbrook003@regis.edu

MS Student in Environmental Biology, Department of Biology

Regis University

November 2019

Section 1: Abstract

Wetland conservation in open space areas is crucial for the vital ecological services provided by these ecosystems. Wetlands provide habitat to a wide variety of flora and fauna, but due to anthropogenically-induced disturbances these systems have become increasingly threatened. Kettle ponds are a rare type of wetland, because of the glacial mechanism that forms them and the overall minute number of ponds that exist. In Colorado, kettle ponds provide habitat to various endemic, sensitive species and increased human activity in close proximity to these ponds threatens the ecological functions the ponds inherently provide. In Boulder County Parks and Open Space (BCPOS) properties, there are several areas encompassing kettle ponds that have been subjected to anthropogenically-induced disturbances. Polluted runoff from agriculture and raising livestock are the most predominate types of human disturbance near BCPOS. I propose to conduct a two-phase study that includes analyzing the impacts on vegetation community composition of kettle ponds subjected to livestock grazing and a corresponding laboratory-based experiment that will quantify the effects of herbicides on odonates found in kettle ponds. The results of my study will provide a greater understanding of how human activity affects these systems. The conclusions drawn from my results will aid BCPOS land managers' target areas in need of restoration and/or protection.

Section 2: Objectives, Hypotheses, Anticipated Value, Literature Review

Objective & Specific Aims

The objective of this study is to quantify how disturbance influences ecological function in kettle ponds on Boulder County Parks and Open Space properties. Specifically, the study will examine how livestock grazing and agricultural runoff influence the habitat kettle ponds provide for wetland flora and fauna. I propose (i) to measure how grazer exclusion influences wetland plant species abundance, richness and cover and (ii) to conduct an experiment to ascertain how mortality and growth rate of kettle pond fauna populations respond to herbicides, a common feature of agricultural effluent. The results of this two-phase study will allow me to document whether the ecosystem services performed by BCPOS kettle ponds can be maintained when subject to livestock grazing and herbicides.

Questions & Hypotheses

Q1: How does livestock grazing affect kettle pond vegetation community composition?

H1: Because livestock grazing decreases vegetation biomass and introduces undesired seeds, kettle ponds with active grazing will exhibit decreased plant species abundance, compared to kettle ponds without active grazing.

Q2: How does polluted agricultural runoff affect a kettle pond's ability to provide wildlife habitat for damselfly populations?

H2: Agricultural runoff containing herbicides will decrease habitat quality which will subsequently decrease survivorship and growth rate, resulting in damselfly population declines.

Anticipated Value

The findings from this study will benefit BCPOS land managers when targeting critical habitats to restore and/or protect. Furthermore, these results will inform both the public about the ecological impacts of agriculture and livestock grazing on kettle ponds, as well as surrounding landowners about how they might manage their land to protect these ecosystems, such as reducing herbicide use or moving areas where livestock herds graze.

Literature Review

Wetlands provide a multitude of important ecological functions, including providing habitat for a wide variety of flora and fauna, and filtering, cleaning and storing water (Cronk & Mitsch, 1992). Today more than half of the world's original wetlands no longer exist. Although natural stressors such as erosion and drought can decrease wetland extent, anthropogenic-induced disturbances, such as agriculture and livestock grazing, have reduced wetland coverage at a much faster rate (Keddy, 2010). Consequently, these stressors have compromised the ecosystem functions provided by these wetlands (Lee Foote & Rice Hornung, 2005). Kettle pond wetland systems have been particularly threatened by human disturbances because they are rare on the landscape and their watersheds have been developed to support agriculture and livestock (Roman, Barrett, & Portnoy, 2001).

Kettle ponds are unique wetland ecosystems that provide all wetland services but have been encumbered by human-induced disturbance (McMurry et al., 2016). Kettle ponds, also referred to as potholes (McMurry et al., 2016), are rare in the landscape because of the mechanism that forms them (Corti, 2012). Ice chunks left behind by retreating glaciers became partially buried in sediment and melted slowly, leaving behind depressions which formed ponds (or lakes) when filled with water (Smith et al., 2000). Many kettle ponds lie along the line of glacial retreat and range in size from 15 feet in diameter to several miles long (Corti, 2012). The majority of these systems are found in the Northeast U.S., because this region experienced glacial activity during the last glacial maximum, but kettle ponds can be found anywhere glacial retreat has occurred, including the glacier-covered mountain regions of Colorado (Smith et al., 2000). In Colorado, kettle ponds provide habitat to many endemic species, including the Hudsonian emerald dragonfly (*Somatochlora hudsonica*) (Packauskas, 2005), tiger salamander (*Ambystoma tigrinum*), mountain toad (*Bufo boreas*) and western chorus frog (*Pseudacris triseriata*), as well as a variety of avifauna and native vegetation; all of these species' populations are decreasing in size (Laubhan, 2004).

Human-induced disturbance events, such as livestock grazing and agricultural runoff, trigger strong changes in the environment (Newman, 2019) and have significantly affected kettle ponds by reducing habitat suitability and water quality. When livestock graze near kettle ponds they directly impact the ecosystem by consuming plant biomass, trampling plants, introducing and dispersing undesired seeds, as well as increasing nutrient inputs and bacterial contamination from dung/urine (Lee Foote & Rice Hornung 2005). Many studies found that livestock grazing near kettle ponds negatively affected water quality, plant species richness, abundance and coverage (Jultila, 1999; Champion et al., 2001; Middleton, 2002; Garnett et al., 2000; Grace & Ford, 1996; Jansen & Healey, 2003). Livestock grazing effects on kettle pond fauna populations are not as significant as the impacts of agricultural runoff contaminated with herbicides (Sura et al., 2012). Herbicides, most frequently glyphosate, but also atrazine and chlorpyrifos, tend to increase in concentration in kettle ponds near cultivated land (McMurry et al., 2016). Although glyphosate generally poses no serious ecosystem risk when applied as directed, it can become more concentrated in runoff from land where it is applied (Imfeld et al., 2013). For example, Relyea (2005) found that amphibian abundance and algal cover decreased even when glyphosate was applied at recommended rates. This occurred because the chemicals were applied directly to mesocosms filled with aquatic fauna (Relyea, 2005), which parallels the effects of increased herbicidal concentrations in agricultural runoff (Imfeld et al., 2013) demonstrating the detrimental impacts glyphosate has on wetland fauna.

Several areas within BCPOS contain kettle ponds, the majority of which experience anthropogenic-induced disturbance. There is little research on how kettle ponds in Colorado have been impacted by these disturbances, which could be due to their rarity in the region (Koehler & Thomas, 2000). The results of my study will provide a greater understanding of how human activity affects these rare and unique systems. The conclusions drawn from my results will aid BCPOS land managers' target areas in need of restoration and/or protection.

Section 3: Methods and Potential Negative Impacts

Methods

Study Site: This experiment will focus on four kettle ponds located in the western portion of BCPOS roughly 2.9 kilometers (km) east of the Coney Flats trailhead (Figure 1). These ponds lie on both the north and south sides of the road and are adjacent to several cattle ranches that graze the area. The two most western ponds, Pond 1 and 2 depicted in Location 1, will be subjected to the livestock grazing treatment (Figure 1). The two most eastern ponds, Pond 3 and 4 depicted in Location 2, will be used for odonate larvae collection (Figure1).

(i) *Quantifying the effects of livestock grazing on kettle ponds*

Starting in the spring of 2020 after snowmelt, I will conduct a before/after, control/impact (BACI) experiment to determine how livestock grazing influences vegetation community composition in kettle ponds. Following Schooler et al. (2006), I will generate thirty random $1m_2$ plots per pond, stored on a

GPS unit, within a 50-meter radius from pond center to survey for baseline vegetation cover and richness at kettle ponds 1 and 2 (Figure 1) prior to experimental intervention. This vegetation survey method will also be used throughout the experimental period. In early June 2020, after the baseline vegetation survey is completed, safety and silt fencing will be placed around the perimeter of Pond 1 to exclude grazing cattle from nearby ranches while Pond 2 will remain open to grazing. Vegetation surveys will be conducted bi-weekly on both Pond 1 and 2 over a 5-month period to examine how wetland vegetation near kettle ponds changes over time. After the collection period has concluded, I will log all measurements in tidy format. I will conduct a data analysis using general linear regression models (GLM) and subsequent analysis of variance (ANOVA) to test whether livestock grazing has caused a change in plant abundance, richness, and vegetation cover.

(ii) *Quantifying the effects of agricultural pollution on kettle ponds*

Starting in June 2020 (larval emergence), I will conduct a laboratory-based dose-response experiment to quantify the effects of polluted agricultural runoff on the mortality and growth rate of a common odonate, *Coenagrion resolutum* (taiga bluet). *C. resolutum* was selected as the focal species because it is widely distributed throughout high-elevation wetland areas (Cooper, 2014). Using a D-frame net, I will collect sweep samples along the pond edges of Pond 3 and 4 (Figure 1), *C. resolutum*'s preferred area for laying eggs. I will place the bulk sweep samples in 5-gallon polyethylene buckets under pond water and transport them immediately to the laboratory at Regis University to prevent desiccation. I will sort *C. resolutum* specimens out of bulk sweep samples and record body length and weight before placing them in individual wells of 30-well counting trays. Following Relyea (2005), on the first day of the 7-day experimental period I will place two *C. resolutum* specimens in each of the 60 400-mL polyethylene containers filled with 380-mL of deionized water containing mosquito larvae for nourishment. 50 properly labeled sample containers will serve as the experimental samples and the other 10 samples will be for the control. To simulate highly concentrated agricultural runoff, I will use five incremental application rates of glyphosate (Roundup), 0.112, 0.122 ,0.132, 0.142 0.152 μg glyphosate/L, each replicated ten times for a total of 50 experimental units, and deposit the corresponding glyphosate concentration via 10-mL plastic graduated cylinder into each individual container immediately after all specimens are added to the sample containers (Relyea, 2005). At the experiment termination, I will remove the specimens from the sample containers and record the number of surviving *C. resolutum*, as well as body length and weight. After all measurements have been recorded, I will use an ANOVA to test whether or not herbicide dosing increases mortality and growth rates of aquatic organisms. I will also generate a comparison model between the incremental doses to analyze whether mortality and growth rates differ by concentration.

Potential Negative Impacts

The experimental kettle pond (Pond 1) will experience limited impacts, in that it will not be grazed for the 5-month period. The odonates subjected to the herbicide treatment will perish, however the focal species, *C. resolutum*, has healthy population numbers, and therefore incidental takes are warranted.

Section 4: Timeline and Budget

Timeline

Dates	*Activities*	*Deliverables*
May 2020 Start	• Install safety and silt fencing	• Protection against further external factors
Early May 2020 – Early June 2020	• Kettle pond baseline observations and measurements • Remove safety and silt fencing from Pond 2	• Reference information for after experimental phase • Record grazing treatment observations
Early June 2020	• Collect odonate specimens & prepare herbicide treatment experiment	• Start herbicide treatment experiment observations
Mid-Late June 2020 – Mid-November 2020	• *Herbicide treatment* – Record odonate mortalities (between Pond 1 & 2 samples) • *Grazing pond* – record plant species abundance, richness & coverage	• Herbicide treatment experiment termination after 7 days • species abundance, richness and coverage measurements for experimental and control kettle ponds
Early December 2020 – Early January 2021	• Compilation of measurements taken and data analysis • Begin drafting report	• Analysis for ecological function of kettle ponds • Report draft
Early January 2021 – Mid-January 2021	• Revise report draft and finish report	• Final report

Budget

Item	*Justification*	*Cost, unit (Source)*	*Quantity*	*Total Cost*
GPS – Garmin GPSMAP 64s	Mapping vegetation survey plots	$199.97 (Regis)	1	$0
4 ft. x 100 ft. Orange Safety Barrier Fencing	Protection & boundary markers	$29.97/roll (Home Depot)	3	$89.91
1/4 in. x 3 ft. x 50 ft. Black Heavy-Duty Dot Silt Fence Fabric	Protection (hydrological) & boundary markers	$14.57/roll (Home Depot)	6	$87.42
Fuel Gas	Travel to & from site	$6.70 – round trip (estimated through Fuel Economy Trip Calculator	13 (trips)	$150.00
$1m^2$ Quadrats PVC	Vegetation % cover & species abundance/richness measurements	PVC quadrats (Regis)	1	$0
Waders	Wading into ponds for measurements and collection	$39.99 (Regis)	1	$0
Notebook – Rite in the Rain	Recording measurements	$5.99 (Amazon)	1	$5.99
Herbicide - Glyphosate	Odonate dosing	$22.69/gallon (Home Depot)	1	$22.69
5-gal polyethylene bucket	Bulk sample storage	$8.91/3-pack (Regis)	1	$0
400 mL polyethylene containers	Storing samples during experiment	$1.00/2-pack (Dollar Tree)	30	$30.00
Mosquito larvae	Nourishment for odonate	$75.95/500-pack (AquacultureStore)	1	$75.95
10 mL plastic graduated cylinder	Application of glyphosate	$2.65 (Regis)	1	$0
30-well counting tray	Odonate storage in preparation of lab experiment	$11.50 (Regis)	1	$0
Centimeter ruler	Measure odonate body length	$1.99 (Regis)	1	$0
Digital scale (gram)	Measure odonate body weight	$34.99 (Regis)	1	$0
TOTAL PROPOSAL REQUEST				$461.96
Regis University has agreed to allow the use laboratory facilities, as well generously donating multiple pieces of equipment.				

Selected Kettle Ponds Map

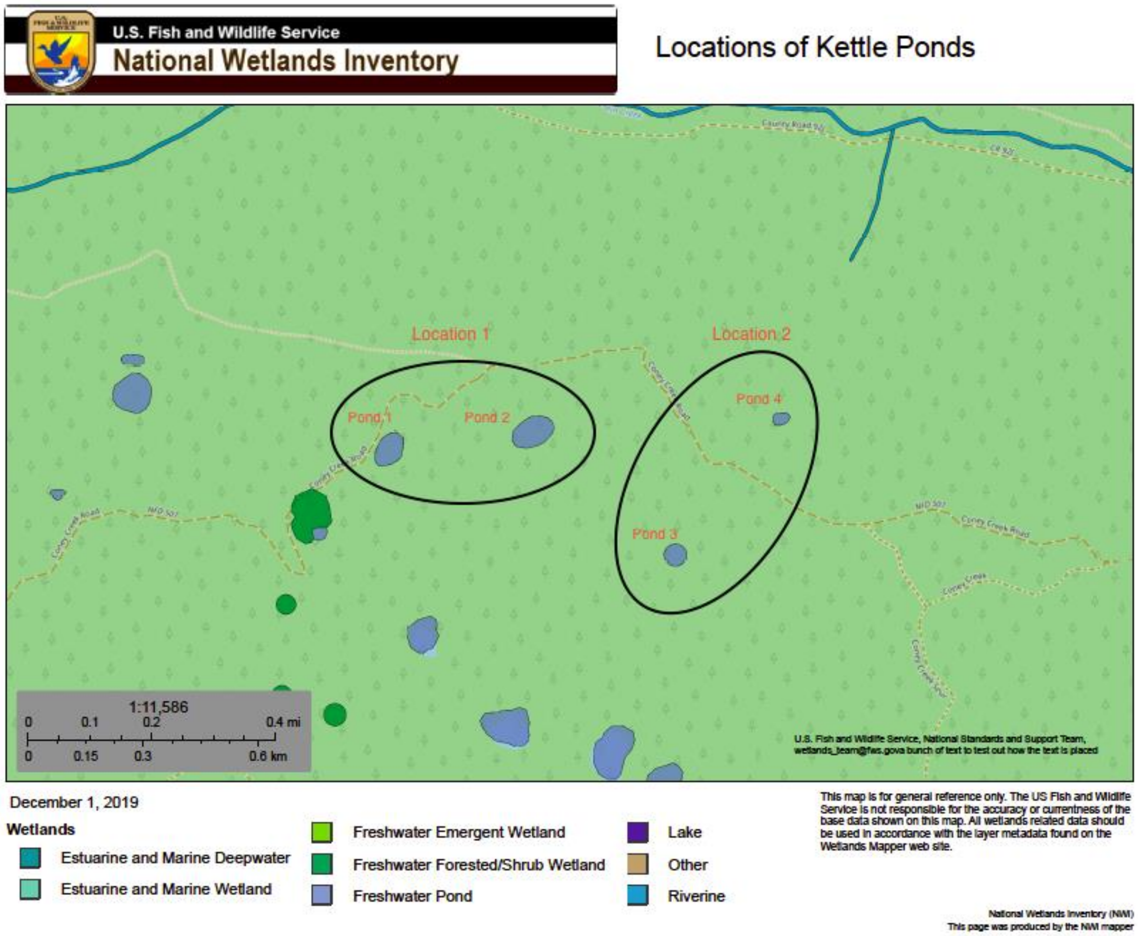

Figure 1: Displays kettle pond (experimental & control ponds) locations in BCPOS. Location 1 depicts location of livestock grazing pond to the west and control pond to the east. Pond 1 is subjected to treatment and Pond 2 is the control. Location 2 depicts Pond 3 and 4 used for specimen collection.

Chase Westbrook

3168 W 14th Ave • Denver CO 80204 • (607)765-0935 • westbrookcww@gmail.com

Profile

Driven student of science with extensive experience in the environmental field. Highly motivated and strong leader willing to go above and beyond to complete any objective. Strong developing knowledge of ecological systems and the comprehensive research techniques that encompass the field.

Education

Le Moyne College, Syracuse, NY
B.S. in Environmental Science Systems, 2013-2017

Regis University, Denver, CO
M.S. in Environmental Biology, 2018-2019

Research/Projects

Research project on genetic sequencing of common microbial species (presented at Phycological Society of America Conference)

The Diversity of House Gutter Eukaryotic Microorganisms

- Assessing the microbial diversity of various habitats leads to the discovery of new lineages and recent studies have turned their focus to the microbial diversity of houses. As a part of the citizen science project, Le Moyne College performed a study to determine the eukaryotic diversity of house gutter communities in Raleigh, NC.

Research of the ecology of the Everglades

- Multiple locations surveyed in the Everglades including several national parks. Observing and analyzing the ecology of the numerous habitats

Research of the effects of vines on trees in temperate regions

- Investigation of possible relationship between vine species effect on native tree species of a northwest U.S temperate forest

Work Experience

Shumaker Engineering & Land Surveying, Binghamton, NY 13901

- Wetlands Biologist/Environmental Scientist, April 2018-July 2019
- Wetland field delineation in accordance with USACE 1987 Manual and Northeast Regional Supplement, environmental field screening of ecological communities to identify presence of species habitat, documenting stream characteristics and flagging Ordinary High Water elevation, preparation of permit application packages for wetland and stream impacts, analyzing data and preparing figures using GIS (ARC GIS), preparation of technical reports under the direction of senior staff, assess project-related impacts, asbestos inspections, storm water management

Delta Engineers, Architects, & Land Surveyors, Endwell, NY 13760

- Environmental Intern, May 2016-August 2016
- Environmental project design e.g. remediation design; monitoring services and data analysis including that of asbestos and lead-based paint control; sampling and evaluating water and air quality

Binghamton University, Binghamton, NY 13902

- Student assistant in the Harpur College Dean’s office, May 2015-August 2015
- Aiding dean and assistant deans with daily agenda; budget tables/graphs, study abroad forms, grant forms, teacher assignment forms, etc.

Training/Certifications

- Army Corps of Engineers Wetland Delineation/Regional Supplement/Waters of the United States Training
- Certificate of Erosion & Sediment Control Training
- New York State Asbestos Investigator Certification

Skills

- Geographic Information System
- R (programming language)
- Microsoft Office

References

Champion, P.D., Beadel, S.M. & Dugdale, T.M. (2001). Turf communities of Lake Whangape and experimental assessment of some potential management techniques 2001. *Science for Conservation*, *186*, 54.

Cooper, A. (2014). *Dragonflies of the Colorado Front Range: a photographic guide*. Boulder, CO: Boulder County Nature Association.

Corti, G., Cocco, S., Basili, M., Cioci, C., Warburton, J. & Agnelli, A. (2012). Soil formation in kettle holes from high altitudes in central Apennines, Italy. *Geoderma*, *170*, 280-294.

Cronk, J.K. & Mitsch, W.J. (1992). Creation and restoration of wetlands: some design consideration for ecological engineering. *Soil Restoration, 17*, 217-259.

Garnett, M.H., Ineson, P. & Stevenson, A.C. (2000). Effects of burning and grazing on carbon sequestration in a Pennine blanket bog, UK. *The Holocene, 10*(6), 729-736.

Grace, J.B. & Ford, M.A. (1996). The potential impact of herbivores on the susceptibility of the marsh plant *Sagittaria lancifolia* to saltwater intrusion in coastal wetlands. *Estuaries, 19*(1), 13-20.

Imfeld, G., Lefrancq, M., Maillard, E. & Payraudeau, S. (2013). Transport and attenuation of dissolved glyphosate and AMPA in a stormwater wetland. *Chemosphere, 90*(4), 1333-1339.

Jansen, A. & Healey, M. (2003). Frog communities and wetland condition: relationships with grazing by domestic livestock along an Australian floodplain river. *Biological Conservation, 109*(2), 207-219.

Jutila, H. (1999). Effect of grazing on the vegetation of shore meadows along the Bothnian Sea, Finland. *Plant Ecology, 140*(1), 77-88.

Keddy P.A. (2010). *Wetland Ecology: Principles and Conservation*. New York: Cambridge University Press.

Koehler, D.A. & Thomas, A.E. (2000). Managing for enhancement of riparian and wetland areas of the Western United States: an annotated bibliography. General Technical Report RMRS-GTR-54. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT.

Laubhan, M.K. (2004). Variation in hydrology, soils, and vegetation of natural palustrine wetlands among geologic provinces. In: *Wetland and riparian areas of the Intermountain West: ecology and management*, 23-51, University of Texas Press.

Lee Foote, A. & Rice Hornung, C.L. (2005). Odonates as biological indicators of grazing effects on Canadian prairie wetlands. *Ecological Entomology, 30*(3), 273-283.

Middleton, B. (2002). Nonequilibrium dynamics of sedge meadows grazed by cattle in southern Wisconsin. *Plant Ecology, 161*(1), 89-110.

McMurry, S. T., Belden, J. B., Smith, L. M., Morrison, S. A., Daniel, D. W., Euliss, et al. (2016). Land use effects on pesticides in sediments of prairie pothole wetlands in North and South Dakota. *Science of The Total Environment, 565*, 682-689.

Newman, E.A. (2019). Disturbance ecology in the Anthropocene. *Frontiers in Ecology and Evolution, 7*(147), 1-6.

Packauskas, R. J. (2005). Hudsonian emerald dragonfly (Somatochlora hudsonica): A technical conservation assessment. USDA Forest Service, Rocky Mountain Region.

Relyea, R.A. (2005). The impact of insecticides and herbicides on the biodiversity and productivity of aquatic communities. *Ecological Applications, 15*(2), 618-627.

Roman, C. T., Barrett, N. E. & Portnoy, J. W. (2001). Aquatic vegetation and trophic condition of Cape Cod (Massachusetts, U.S.A.) kettle ponds. *Hydrobiologia*, *443*, 31-42.

Schooler, S.S., McEvoy, P.B. & Coombs, E.M. (2006). Negative per capita effects of purple loosestrife and reed canary grass on plant diversity of wetland communities. *Diversity Distributions, 12*(4), 351-363.

Smith, R.J., Hines, A., Richmond, S., Merrick, M., Drew, A. & Fargo, R. (2000). Altitudinal variation in body size and population density of Nicrophorus investigator (Coleoptera: Silphidae). *Environmental Entomology, 29*(2), 290-298.

Sura, S., Waiser, M., Tumber, V., Lawrence, J.R., Cessna, A.J. & Glozier, N. (2012). Effects of glyphosate and two herbicide mixtures on microbial communities in prairie wetland ecosystems: a mesocosm approach. *Journal of Environment Quality, 41*(3), 732-743.

CHAPTER 3. JOURNAL MANUSCRIPT

Status of Mountain Plover (*Charadrius montanus*) Populations in South Park, CO: Post-Conservation

Abstract

The mountain plover (*Charadrius montanus*) is a small, insectivorous grassland bird that occurs in the Western United States. Populations of mountain plover have historically experienced significant declines, primarily due to habitat loss caused by human land conversion for development. In South Park, Colorado, the Bureau of Land Management maintains an area that provides habitat to approximately 2,300 breeding individuals of mountain plover. This group of birds represents a significant portion of the continental population of mountain plover, making the land vitally important to protect for the survival of the species. BLM conducts mountain plover surveys on a triennial basis and have been investigating statistical methodologies to assess the population health of mountain plover utilizing their survey data. Previously, BLM researchers employed a frequentist approach to estimate the probability of observing mountain plover as a proxy for site occupancy, a common method in bird surveys. In this study, I analyzed the same survey data using several Bayesian models that communicate uncertainty via posterior distributions on important parameters of interest.

Introduction

Bird surveys are a common tool used to determine health of avian populations over time. While bird surveys typically provide information on trends in abundance, they also provide data on reproduction, migration and habitat preference (McCaffery & Ruthrauff, 2004). Occupancy counts and nesting activity are both frequent targets of monitoring and management efforts because they act as proxies for breeding activity (Mayfield 1961; Klett and Johnson 1982). Effective monitoring requires that we understand and account for potential sources of bias, including imperfect detection of birds and their nests (Burnham, 1981; Anderson, 2001; Smith et al., 2009), in the models we use to analyze occurrence data. Imperfect detection is a widely recognized problem in ecological surveys and when species detection is imperfect, surveyors may classify a site as unoccupied even when the species is present at the site (Guillera-Arroita et al., 2014). Imperfect detection is caused by a number of factors including variability in detection and behavior among individuals, where traits such as size and color may affect the ability of an observer to detect an organism in its natural habitat (Ringelman 2014). There are also site- and survey-specific factors which have been shown to influence detectability as well, such as habitat structure at a survey site or inclement weather that impede an observer's ability to detect all individuals present (Iknayan et al., 2014). Failure to account for imperfect detection of individuals biases estimates of true occupancy, most commonly population underestimates which can potentially lead to erroneous conclusions about population status and habitat requirements (Kéry & Schmidt, 2008). It is important to account for detection bias when drawing inferences from surveys and monitoring, especially when sampling highly mobile groups like threatened ground-dwelling bird species that shelter for the majority of the day, increasing the likelihood of non-detection when a site is occupied (Augustine & Derner, 2012).

Because ground-dwelling bird species spend the majority of their daily activity budget on the ground surface, they have become increasingly threatened by anthropogenic stressors throughout the past century. Ground-dwelling avian species, also known as waders but more commonly described as shorebirds, encompass 210 identified species globally (Stroud et al., 2006). Most ground-dwelling birds occur in coastal regions but may occupy inland areas that have suitable habitat for populations to feed, reproduce and access shelter (Skrade & Dinsmore, 2010). Ground-dwelling birds forage and breed in both wetland and upland habitats, the latter of which are used for courtship and mating, nesting, and rearing young (Stroud et al., 2006).

Over the last century, land conversion for development has destroyed a large portion of the habitat of many ground-dwelling bird species (Augustine & Derner, 2012), resulting in steep population declines (U.S. Fish and Wildlife Service, 1999). In addition, numerous ground-dwelling bird species are or have been on the verge of extinction due to resource extraction activities, such as oil and gas drilling. The high frequency of these exploits in terrestrial regions results in the widespread destruction of suitable habitat (Carter et al., 2000). Resource extraction not only destroys suitable habitat for ground-dwelling birds, but also disrupts natural activity patterns by reducing food availability and mating opportunities (Augustine & Derner, 2012). Loss of habitat to human development threatens many species of ground-dwelling avian species, including the mountain plover (*Charadrius montanus*).

The mountain plover is a member of the largest shorebird order, *Charadriinae*, with approximately 66 species of plover identified globally (Skrade & Dinsmore, 2010). The mountain plover is a small (21-23 cm length), insectivorous grassland bird that occurs in the Western U.S. Mountain plover share common morphological traits with the extant plover group, distinguished by relatively short rostrums, long legs, and feeding behavior characterized by run-stop-tilt forward action on areas of open sand, mud, shingle, bare earth or short grass (Knopf & Wunder, 2006). Mountain plover are associated with shortgrass prairie, which is often dominated by blue grama (*Bouteloua gracilis)* and buffalo grasses (*Buchloe dactyloides*) (Graul, 1975). It primarily nests on disturbed and grazed habitat, and markedly prefers to nest on prairie dog

(*Cynomys sp.)* colonies throughout much of its range (Knowles et al., 1982). Mountain plover breeding habitat occurs primarily in Colorado, Wyoming, Montana, and New Mexico, with smaller populations breeding in Kansas, Utah, Nebraska, and Texas. Migration begins in late summer (July-September) as plovers relocate to wintering grounds in the Southwestern U.S. (primarily California) and Northern Mexico. In the Southwestern U.S., habitats supporting mountain plover populations are threatened by land use and land cover change.

Habitat destruction/modification, agricultural practices, management of domestic livestock, and loss of native herbivores have resulted in declines of mountain plover populations (U.S. Fish & Wildlife Service, 1999). Populations of mountain plover declined by 63% from 1966 to 1996, according to Breeding Bird Survey data (U.S. Fish & Wildlife Service, 1999). In 1999, the North American population was estimated to be between 8,000 and 10,000 individuals (U.S. Fish & Wildlife Service, 1999). In recent decades, similar population declines have also been observed in other plover species including the piping plover (*Charadrius melodus*), hooded plover (*Thinornis rubricollis*) and killdeer (*Charadrius vociferous*) (Augustine & Derner, 2012). Like other threatened plover species, mountain plovers are highly susceptible to changing environmental conditions because their distribution is restricted and population sizes are small (Isaksson, Wallander & Larsson, 2007).

In response to the mountain plover decline, the most severe of any native grassland bird in North America, the U.S. Fish and Wildlife Service (USFWS) listed the mountain plover as a Threatened Species under the Endangered Species Act in 1999. Correspondingly, the U.S. Forest Service and the Bureau of Land Management (BLM) labeled it a Sensitive Species, and the Colorado Division of Wildlife listed it as a Species of Special Concern. Surveys in the last two decades suggest that the total population of mountain plovers may have increased. Wunder et al.

(2003) reported the total population to be between 10,000 and 12,000 individuals, whereas Tipton et al. (2009) suggested that the estimate be revised to 15,000-20,000 birds. In May 2011, the USFWS de-listed the mountain plover as a Threatened Species, stating that "threats to the mountain plover identified in the proposed rule were not as significant as previously believed." (U.S. Fish & Wildlife Service, 2011). The agency concluded that the plover population would not be imperiled in the foreseeable future.

Conservation efforts in South Park, the focal area of this study, are important because the birds that breed in South Park (approximately 2,300 individuals) represent a significant portion of the continental population of mountain plovers (Wunder et al. 2003). This area appears to be a stronghold for mountain plovers during the breeding season (Wunder et al. 2003). Although Oyler-McCance et al. (2005) found no evidence of genetic differentiation among mountain plover populations range-wide, contradicting the findings of Wunder et al. (2003). They suggest that juvenile female birds may form pair bonds during migration or at wintering grounds and establish breeding territories at places other than their natal habitat (Oyler-McCance et al., 2005). Because this population of mountain plover is of significant importance, researchers at the BLM established a long-term monitoring effort to investigate trends in mountain plover occupancy, critical data needed for conservation management plans. The researchers addressed this effort with a frequentist model approach and found that the probability of detecting an individual was 0.152 in 2014 and 0.102 in 2017 (BLM, 2017). Unlike the frequentist framework employed by the BLM, a Bayesian approach may be the optimal choice because Bayesian models employ a rigorous probabilistic framework to represent the uncertainty in any event or hypothesis. Bayesian analyses are valuable for the ongoing BLM monitoring effort because newly collected data may be used to update the distributions for parameters, like occupancy probability, that are used to make important conservation decisions.

In order to construct a model that better predicts plover population estimates and future trends, the factors that strongly influence plover population fluctuations and cause uncertainties in occupancy estimates must be identified. Factors such as imperfect detection and other human biases frequently underestimate occupancy. By analyzing longitudinal mountain plover survey data in a Bayesian framework, I plan to assess how accounting for imperfect detection affects occupancy estimates. Because a frequentist framework does not account for these uncertainties, I expect transitioning to a Bayesian approach for modeling mountain plover occupancy will result in a more accurate estimate of both the probabilities of detection and occupancy. I will also examine how seasonal and daily survey timing influence mountain plover detection, because of the variability in detectability due to behavioral trends of mountain plover (Augustine & Derner, 2012). Mountain plover typically migrate in the late summer and seek shelter during the daytime, therefore causing the detection of individuals less likely later in the season and later in the day. Because this timing factor was not addressed in the previous frequentist approach, I expect a Bayesian model that includes these factors will result in a more precise estimate of observation probability. Furthermore, due to their behavioral trends I am expecting that the probability of observing an individual will have a negative relationship with both later seasonal (Julian day) and daily (decimal time) survey timing. Ground-dwelling birds, and specifically that of my focal species the mountain plover are of particular concern due to constant anthropogenic ground disturbance over the past several decades. Assessing how imperfect detection and survey timing influence mountain plover occupancy will help land managers analyze data from surveys in a way that will improve conservation planning for mountain plover and other species of concern.

<u>Methods</u>

Study Area:

The study area was located on public land managed by the BLM in South Park, Park County, Colorado. The BLM manages approximately 13% (11,000 ha) of potential plover habitat in South Park (Grunau & Wunder, 2001). The dominant habitat in this area is montane grassland interspersed with pockets of shrubs (e.g., rabbitbrush [*Chrysothamnus spp*.]) and trees including aspen (*Populus tremuloides*), ponderosa pine (*Pinus ponderosa*), and bristlecone pine (*Pinus longaeva*). Elevation averaged 2850 m over the relatively uniform study area. South Park is drained by the South Platte River and tributaries of the Tarryall Creek. The climate in this area is characterized by short, cool summers and long, cold winters, with approximately 26 cm precipitation annually (Wunder et al., 2003). Average minimum and maximum temperatures are -12.8 and 20.6 ℃, respectively (Grunau & Wunder, 2001).

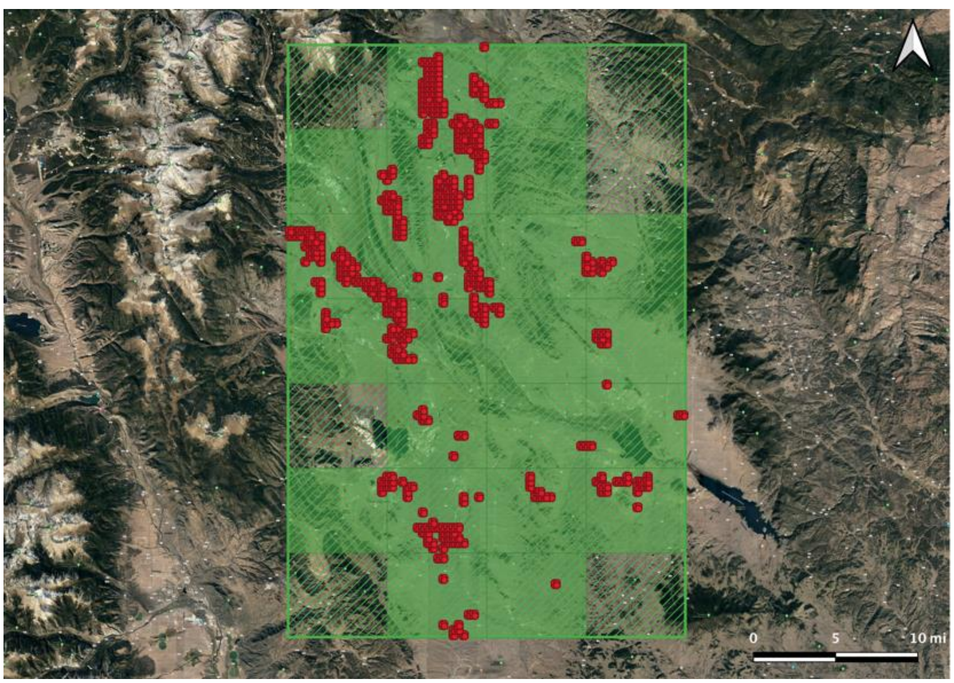

Figure 1. Depicts BLM managed area located in South Park, Park County, CO consisting of Mountain Plover grassland habitat. Points (red) show areas where surveys were previously conducted.

Mountain Plover Survey:

BLM technicians used ArcGIS 10.3 to overlay a grid of 500×500 m survey plots where BLM land intersected mountain plover habitat as identified by the Colorado Natural Heritage Program in 2001 (Grunau & Wunder, 2001). Technicians surveyed 271 randomly selected plots out of a possible 414 plots during May, June, and July 2017, conducting surveys between 0540 and 1030 MST. Previous survey years included 2012 and 2014, where 220 and 414 randomly selected plots, respectively were surveyed. In order to implement a Bayesian framework, I used only the sites that were surveyed in all three of the survey years (2012, 2014, 2017), which resulted in a total of 107 replicate sites. Survey design was based on Tipton et al. (2009), with a single observer walking transects placed to allow a view of all parts of the plot within 125 m (Figure 2). If a portion of the survey plot fell outside of BLM land, observers walked the boundary and used binoculars to observe as much of the plot as possible. Technicians recorded plover presence/non-detection, survey start and end times, number of adults observed, presence/non-detection of nests, number of eggs, number of chicks, weather conditions, temperature, and wind speed using the Beaufort Wind Scale. When a plover was detected during the survey, the survey ended at that time. To determine the number of randomly selected plots to survey the subsequent sampling period, BLM researchers previously conducted a data analysis that encompassed a frequentist approach for modeling mountain plover occurrence. This was done utilizing the R programing language to conduct a power analysis to determine the number of plots needed to detect a 5% decrease or 10% increase in occupied blocks within each of the surveyed plots. The researchers concluded that for the 2017 survey the estimated number of plots needed to detect a 5% decrease in occupied blocks would be 253. The estimated number of plots needed to detect a 10% increase in occupied blocks would be 243, therefore the resultant sample size the researchers decided to use was 253 sample plots.

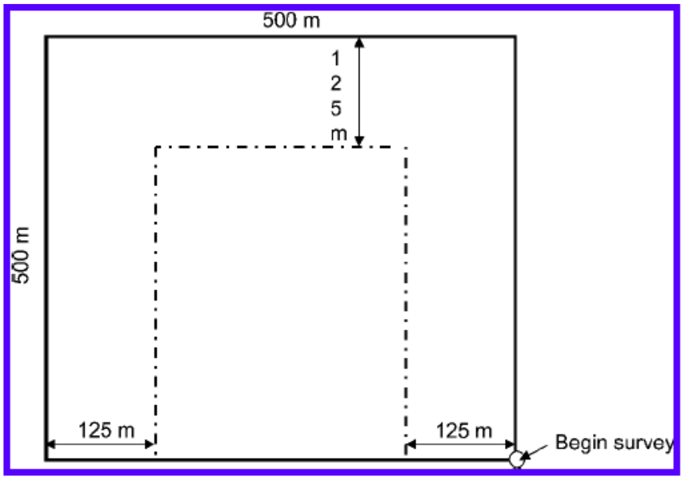

Figure 2. Diagram of transect walked by observers during surveys conducted on randomly selected 500×500m plots (Tipton et al., 2009)

Data Analysis:

Rather than use frequentist estimates of occupancy, as the BLM did for the 2017 survey period, I utilized Bayesian methods to fit three models: (1) a simple model of observation probability in each year, (2) a model that assumed observation probability was related to seasonal and daily survey timing in each year, and (3) a model that differentiated occupancy and detection probabilities using sites that were visited in all three years. For the detection/non-detection data to be of practical use in the third of these analyses, the raw survey data was first filtered for sites that were visited in all three survey years (2012, 2014, 2017), resulting in a total of 107 replicate sites. All models were fit using Just Another Gibbs Sampler (JAGS) (Plummer, 2003), aided by the following Rpackages: jagUI, rjags, unmarked and lubridate, with pre- and post-processing in R version 3.6.1 (R Core Team, 2019; Plummer, 2003). JAGS sampled the posterior distribution of the important parameters using three independent Markov Chain Monte Carlo (MCMC) chains that were fit for 10,000 iterations. Because the early values of the chain can be highly dependent on the initial values, the first 5,000 iterations of each chain were discarded as ‘burn-in.’ The last 5,000 iterations were subsequently thinned by 1/5 to ensure that posterior samples were independent. The posterior distributions were sampled and post-processed to calculate summary statistics including means and 95% credible intervals in the MCMCvis package (Youngflesh, 2018). The three models I decided to fit were adapted from examples in Schaub and Kery (2011).

Annual Detection Probabilities (without covariates):

In these models, each year (2012, 2014, 2017) was analyzed separately in Bayesian model framework, with no covariates included and perfect detection assumed. These annual models assumed that whether a mountain plover was observed followed a Bernoulli distribution, with an underlying observation probability (p). Before conducting this study, I had little information on the observation probability of mountain plover, therefore I assumed a uniform prior on the detection probability (p) on the interval from 0 to 1. Since covariates were not included, these models are simple null models that estimate the probability that plover was observed in each year.

Annual Detection Probabilities (covariates included):

After fitting simple null models, I then fit annual models that assessed how probability of observation was influenced by the survey timing covariates Julian day and decimal time. These models again assume perfect detection. As in the null model case, these annual models assumed that whether a mountain plover was observed at site i followed a Bernoulli distribution, with an underlying detection probability (p_i) which was linearly related to the two time covariates

$$\text{logit}(p_i) = \beta_0 + \beta_1 * d_i + \beta_2 * t_i$$

where β_0 represents the baseline log odds of observation (at the earliest date, 05/27, and time, 5:40MST), β_1 represents the effect of day of the year (d), and β_2 represent the effect of time of day. Priors for β_0, β_1, and β_2 were assumed to be normal with mean 0 and precision 0.1. These models allowed me to observe if survey timing affects the probability of observation for mountain plover.

Detection Probabilities & Occupancy Estimates (Multi-Season Model):

The last model created for South Park mountain plover populations included data from sites that were visited during all years of the study. This Bayesian model is particularly important to the analysis, because it separates the probability of site occupancy from detection for South Park mountain plover populations. In order to simplify and allow for integration into other models, I adopted the notation of other Bayesian occupancy models (e.g., Royle & Kéry, 2007; Schaub & Kery, 2011), and let $y_{i,t}$ represent the detection/non-detection of mountain plover on site *i* during survey year *t* whose distribution was Bernoulli with probability of detection $p_t z_i$. $z_{i,}$ is a latent variable that represents the occupancy of site *i* over the six years of the study. z_i also follows a Bernoulli distribution with underlying occupancy probability, ψ:

$$z_{i,} \sim \text{Bernoulli}(\psi)$$

This model assumes imperfect detection and the entire survey period is considered closed during which occupancies do not change. This is not ideal because over the six years of this study, individual plover may have emigrated or immigrated. However, because sites were not visited multiple times in the same year, we cannot differentiate detection and occupancy on an annual basis. We can now derive the, detection probabilities for each survey year (p_1, p_2, p_3) and site occupancy probability (ψ) (R Core Team, 2019).

<u>Results</u>

Annual Observation Probabilities (without covariates):

Probability of observing mountain plover was similar across all survey years although these probabilities slightly declined over the course of the study. Bayesian models of annual observation probability resulted in the following posterior estimates: 0.174 (95% CI: 0.125 – 0.229) in 2012, 0.155 (95% CI: 0.122 – 0.192) in 2014, and 0.133 (95% CI: 0.096 – 0.176) in 2017.

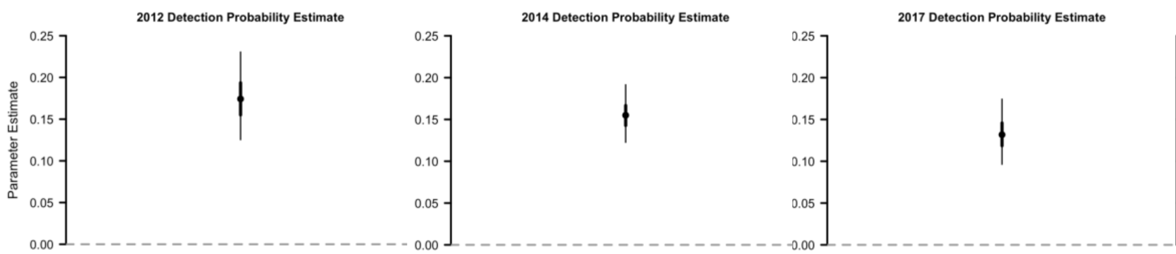

Figure 3. Depicts the observation probabilities based on the simple null estimates of the three survey years (2012, 2014, 2017)

Annual Detection Probabilities (covariate included):

Similar estimates to the simple null model were also found in these models, however the time covariate estimates showed divergence, resulting in the relationship between the observation probability and time covariates to have opposite effects. Annual baseline observation probabilities, the observation probabilities on the earliest date (e.g. 05/27) and time (e.g. 5:40 MST) from the data were: 0.290 (95% CI: 0.137 – 0.489), 0.108 (95% CI: 0.048 – 0.195), 0.167 (95% CI: 0.088 – 0.280) in 2012, 2014 and 2017 respectively. Similar to the models with covariates absent, models including covariates, indicate that observation probabilities were similar across years. Mountain plovers were more likely to be observed later in the season. Posterior estimates indicated that surveyors were more likely to observe mountain plover by

2.24% (95% CI: -2.17 – 7.00), 2.69% (95% CI: 0.53 – 4.78) and 1.35% (95% CI: -2.67 – 5.34) each day later in the season that the survey was conducted in 2012, 2014 and 2017 respectively. Surveyors were 25.5% (95% CI: 7.77 – 38.35) and 28.54% (95% CI: 0.059 – 48.96) less likely to observe plovers later in the day in 2012 and 2017 respectively. However, in 2014, the effect of survey timing was quite uncertain, 5.27% (95% CI: -21.66 – 24.40).

Detection Probabilities & Occupancy Estimates (Multi-Season Model):

This model provides a probability of site occupancy estimate through Bayesian methodology for mountain plover populations, which resulted in a relatively high estimate for the occupancy probability and a decreasing trend in detection probability across the survey years. The probability of a site being occupied is estimated to be $\psi = 0.615$ (95% CI: 0.386 – 0.917) across the six years of the study. The mean detection probability for each year surveyed include: 0.307 (95% CI: 0.159 – 0.489), 0.244 (95% CI: 0.126 – 0.401) and 0.122 (95% CI: 0.049 – 0.230) in 2012, 2014 and 2017 respectively (Figure 4).

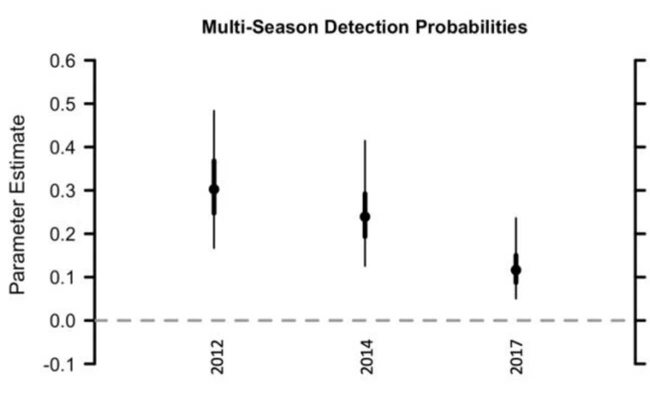

Figure 4. Depicts the detection probabilities based on the combined survey year data (multi-season model)

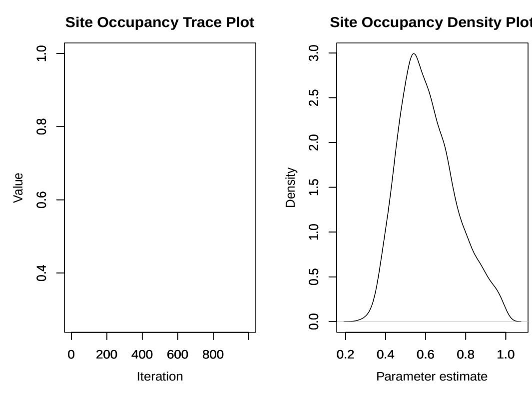

Figure 5. Depicts trace plot (left) and density plot (right) of site occupancy probabilities based on the multi-season model.

Discussion

My results suggest that these occupancy modeling methods are optimal to monitor and address biological questions about mountain plovers over a large space in South Park, Colorado. The observation probabilities based on the simple null models that utilized the detection/non-detection data for each year (2012, 2014, 2017) were estimated to be relatively similar in respect to the degree of change among the survey years, indicating that the probability of observing an individual did not change from year to year. In the models including the time covariates the baseline estimates did not significantly change year to year similar to the simple null model. Although, the probability estimates sampled from time covariates priors did indicate that they do influence the observation probability of mountain plover in a divergent fashion. Julian day was shown to have a positive relationship with the probability of observing an individual, while decimal time had the opposite, negative effect, meaning mountain plover were more likely to be observed earlier in the day and later in the season. When I accounted for imperfect detection in sites that were surveyed in all three years, site occupancy for mountain plover was estimated to be much higher ($\psi = 0.615$) than other mountain plover occupancy studies (Tipton et al. 2008). The results of the Bayesian models I employed demonstrate that imperfect detection underestimates occupancy probabilities, and also verified that the time covariates do in fact have an effect on the probability of observation as initially hypothesized.

Results from the null models showed that mountain plover observation probability was similar across all survey years, but that there was still a noticeable, albeit slight, decrease over time. Because these changes were small, I infer that mountain plover observation probabilities were relatively stable over the course of the study. Because I employed flat priors in these models, the results from the frequentist approach conducted by the BLM had similar probability estimates to the Bayesian models I employed.

After accounting for survey timing, probabilities of observation also showed no clear trends over time. Nonetheless, one can clearly see that time of sampling does influence whether or not a mountain plover is observed, which may be a useful component to include in future models when predicting the probability of observation at a given site. The estimates from the time covariates showed divergence, where the effect of Julian day or seasonal timing of sampling was not as confident about the direction of timing compared to the effect found among the decimal time-level covariate. These effects are likely due to the fact that mountain plovers tend to seek shelter during day for shade, especially during warmer periods (Augustine & Derner, 2012). Mountain plover migratory patterns could also be a determining factor that are causing the effects observed from these results. Migration typically begins in the late summer (July-September) as plovers relocate to wintering grounds in the Southwestern U.S. and Northern Mexico (BLM, 2017), therefore surveys conducted later in the season were most likely impacted by this annual behavior. These findings were in line with Tipton et al. (2008), where they found that if they modeled detection or occupancy probabilities as a function survey timing (date & time) the resultant estimates were quite variable, indicating a lack of confidence in survey timing. These findings are also important for the interpretation of the observation probabilities, where in the BLM frequentist model reported observation probabilities that did not account for covariate effects when estimating probabilities, when in fact these time covariates do have a known effect. By accounting for these types of effects, we would then receive better, more precise baseline estimates which is important in determining the population trends when a known effect is accounted for. Other studies have shown similar results when determining whether or not detection probabilities or occupancy are affected by site- or plot-level covariates, such as elevation, prairie dog and owl populations (Wunder et al., 2003; Augustine & Derner, 2012; Tipton et al. 2008).

The assumption of perfect detection must still be taken into account, since these first two models for the surveys included this assumption. Most detection/occupancy models account for the impacts of imperfect detection in order to alleviate this source of error when developing models (Iknayan, 2018). This correction is particularly important when using historic data because of the differences in methodology and technology from when data was collected, which can alter the detectability of individuals (Iknayan, 2018). My models utilize relatively flat priors due to the absence of historical data, resulting in similar trends between the BLM frequentist approach and the Bayesian approach I employed. Subsequent annual estimates from these models could have been made more precise through Bayesian updating. Instead of using flat priors in 2014 and 2017, I could have used the posterior from 2012 models as the prior information for the 2014 models, and the 2014 posterior estimates as my priors for the 2017 models. To employ Bayesian updating I would only need to parameterize an appropriate distribution for the prior information based on the posterior results from the prior year. Similar trends and issues involving the assumptions and resultant estimates were also observed in the models that included the time covariates, which could have also benefitted from a Bayesian updating system.

When I used a multi-season site occupancy model assuming imperfect detection, I found that the estimate for occupancy was high. This result should be interpreted cautiously because it assumes that the system was closed over the six-year period (2012-2017). This assumption for this time period is likely untrue. Nevertheless, despite these shortcomings multiple studies

(Guillera-Arroita et al., 2014; Kéry & Schmidt, 2008) show that not accounting for imperfect detection results in an underestimate of mountain plover occupancy that is why it is important to consider potential biases in model development. In addition, this model could have integrated the effects of seasonal and daily survey timing on detection probabilities to better improve the estimates. If these factors were included in this model of imperfect detection, uncertainty the occupancy estimate may become more precise. More importantly, future surveys of mountain plover at South Park should include revisits to the same sites multiple times each year. Doing so will make the closed system assumption more likely to be true and achieve better annual estimates of occupancy. As the survey data sets get longer and if BLM takes my recommendation into account, this general model will be very useful for observing changes in occupancy over time.

Tipton et al. (2008) conducted a similar study on the mountain plover populations in Colorado in 2005 before the BLM collected mountain plover data in South Park. The researchers' survey methods for mountain plover were identical to that of BLM, although they replicated sampling efforts at sites multiple times during their collection periods. Tipton et al. (2008) was unable to control individual survey variability because of the larger number of surveyors used in the study, so the researchers decided to model occupancy as constant and accounted for imperfection detection. Tipton et al. (2008) estimated mountain plover site occupancy in relation to three separate land cover types and determined that mountain plover occupancy was high on sites consisting of prairie dog colony sites ($\psi = 0.50$, 95% CI = 0.36 – 0.64), but much lower on grassland ($\psi = 0.07$, 95% CI = 0.03 – 0.15) and dryland agriculture ($\psi =$ 0.13, 95% CI = 0.07 – 0.23) sites. We can assume that the South Park landscape has similar characteristics to that of the prairie dog colony site in Tipton et al. (2008) research, due the ecological understanding of what biota impact the landscape of the South Park site, which includes a interspersed populations of prairie dog (*Cynomys* spp.) and grazing livestock; a habitat trait favored by mountain plover. Still, the discrepancy between the Tipton et al. (2008) estimate and my estimate is large (~0.12), potentially due to the fact that the South Park mountain plover population is under conservation protection and is much larger than the sites Tipton et al. (2008) utilized for their sample group. One last issue involving the differences between the estimates, could simply be the fact that my model assumes that this population of mountain plover is a closed system, thereby causing high estimates for the occupancy probabilities. However, if the assumption of imperfect assumption is not accounted for this could have led to an underestimate of the population. Even though in this instance where accounting for imperfect detection resulted in a high occupancy probability estimate, it is still vitally important to take into account potential biases such as imperfect detection because underestimates can be detrimental to previously listed and still vulnerable species.

Maintaining and updating population occupancy counts is vital for conservation efforts involving threatened species. The fact that federal and state institutions no longer designate the mountain plover as threatened or endangered, does not necessarily indicate that the species is no longer vulnerable, particularly when statistical modeling that tend to confirm these listings underestimate the population heath of the species under consideration (McGowan et al., 2017). Previously listed species, such as the mountain plover that are still recovering are still vulnerable to population loss, which could result in re-listing (Ferraro, McIntosh & Ospina, 2007). That is why it is important to keep up-to-date population records and implement models that can precisely and accurately predict population trends (Kéry & Schmidt, 2008). Site occupancy models are one of the most optimal methods to determine population health. A frequentist approach in model development is a common framework when analyzing population site occupancy (Guillera-Arroita, 2014), but lacks certain criteria such as degrees of belief and logical support when assessing probabilities of detection as observed in the Tipton et al. (2008) article.

Occurrence records offer the opportunity to broaden our understanding of biological change across taxonomic groups if analyzed using appropriate statistical methods. The occupancy model is flexible and is gaining familiarity with ecologists (Bailey et al. 2014), whereas the Bayesian framework allows for latent states and added complexity through its conditional probabilities (Latimer et al., 2006). I developed three models in a Bayesian framework that are optimal for estimating detection probabilities and more importantly site occupancy for mountain plover located in South Park, CO. In order to improve this model, I suggest BLM surveyors sample the same sites each year, focusing on replicate sampling of sites year by year rather than decreasing the amount of sampling in general as initially proposed. By increasing the number of sites surveyed multiple times, the current Bayesian model I developed for site occupancy will increase in precision and will confirm the closed system assumption, allowing for future inferences of true occupancy. I would also recommend that time covariate factors are taken into account in future analyses, as well as other environmental variables that may have an effect on detection and occupancy probabilities. Special attention should be focused on differing site- and plot-level factors, such as surrounding populations of different wildlife that have known effects on plover detectability and occupancy, as well as the different geomorphic environmental variables (e.g. elevation and aspect of each plot). By taking these recommendations into account for future analyses, one will have a much better model in regard to the precision and accuracy of the estimated detection and occupancy probabilities for mountain plover populations.

References

Anderson, D.R. (2001). The need to get the basics right in wildlife field studies. *Wildlife Society Bulletin, 29*, 1294-1297.

Augustine, D.J. & Derner, J.D. (2012). Disturbance regimes and mountain plover habitat in shortgrass steppe: large herbivore grazing does not substitute for prairie dog grazing or fire. *The Journal of Wildlife Management, 76*(4) 721-728.

Bailey, L.L., MacKenzie, D.I. & Nichols, J.D. (2014). Advances and applications of occupancy models. *Methods in Ecology and Evolution, 5*, 1269-1279.

BLM. (2017). Occurrence of mountain plover in South Park, Park County, Colorado 2017.

Brown, C.R. & M. Bomberger. (2000). Nest spacing in relation to settlement time in colonial cliff swallows. *Animal Behaviour, 59*, 47-55.

Burnham, K.P. (1981). Summarizing remarks: environmental influences. *Studies in Avian Biology, 6*, 324-325.

Carter, M.F., Hunter, W.C., Pashley, D.N. & Rosenberg, K.V. (2000). Setting conservation priorities for landbirds in the United States: the partners in flight approach. *The Auk, 117*, 541-548.

Ferraro, P.J., McIntosh, C. & Ospina, M. (2007). The effectiveness of the US endangered species act: an econometric analysis using matching methods. *Journal of Environmental Economics and Management, 54*(3), 245-261.

Fisher, J.B., Trulio, L.A. & Biging, G.S. (2007). An analysis of spatial clustering and implications for wildlife management: a burrowing owl example. *Environmental Management, 39*, 403-411.

Fiske, I. & Chandler, R. (2011). Unmarked: an r package for fitting hierarchical models of wildlife occurrence and abundance. *J Stat Software, 43*,1-23.

Graul, W.D. (1975). Breeding biology of the mountain plover. *Wilson Bulletin, 87*, 6-31.

Guillera-Arroita, G., Lahoz-Monfort, J.J., MacKenzie, D.I., Wintle, B.A. & McCarthy, M.A. (2014). Ignoring imperfect detection in biological surveys is dangerous: a response to "fitting and interpreting occupancy models." *PLoS ONE, 9*(7), e99571.

Iknayan, K., Tingley, M., Furnas, B. & Beissinger, S. (2014). Detecting diversity: emerging methods to estimate species diversity. *Trends in Ecology and Evolution, 29*(2), 97-106.

Isaksson, D., Wallander, J. & Larsson, M. (2007). Managing predation on ground-nesting birds: the effectiveness of nest exclosures. *Biological Conservation, 136*(1), 136-142.

Kéry, M. & Schmidt, B. (2008). Imperfect detection and its consequences for monitoring for conservation. *Community Ecology, 9*(2), 207-216.

Klett, A.T. & D.H. Johnson. (1982). Variability in nest survival rates and implications to nesting studies. *The Auk*, *99*, 77-87.

Knowles, C.J., Stoner, C.J. & Gieb, S.P. (1982). Selective use of black-tailed prairie dog towns by mountain plovers. *The Condor, 84*, 71-74.

Knopf, F.L. & Wunder, M.B. (2006). Mountain Plover (*Charadrius montanus*). The Birds of North America, No. 211. Cornell Laboratory of Ornithology, Ithaca, NY, USA.

Latimer, A.M., Wu, S., Gelfand, A.E. & Silander Jr, J.A. (2006). Building statistical models to analyze species distributions. *Ecological Applications, 16*, 33-50.

Mayfield, H.F. (1961). Nesting success calculated from exposure. *Wilson Bulletin, 73*, 255-261.

McCaffery, B.J. & Ruthrauff, D.R. (2004). How intensive is intensive enough? Limitations of intensive searching for estimating shorebird nest numbers. *Wader Study Group Bulletin,* *103*, 63-66.

McGowan, C.P., Allan, N., Servoss, J., Hedwall, S. & Wooldridge, B. (2017). Incorporating population viability models into species status assessment and listing decisions under the U.S. Endangered Species Act. *Global Ecology and Conservation, 12*, 119-130.

Oyler-McCance, S.J., St. John, J., Knopf, F.L. & Quinn, T.W. (2005). Population genetic analysis of mountain plover using mitochondrial DNA sequence data. *The Condor, 107*, 353-362.

Patterson, I.J. (1965). Timing and spacing of broods in the black-headed gull *Larus ridibundus*. *Ibis, 107*, 433-459.

Plummer M. (2003). JAGS: A program for analysis of Bayesian graphical models using Gibbs sampling. In: Hornik K, Leisch F, Zeileis A, editors. *Proceedings of the 3rd International Workshop on Distributed Statistical Computing* (DSC 2003). Vienna, Austria. 20-22.

Ringelman, K.M. (2014). Predator foraging behavior and patterns of avian nest success: what can we learn from an agent-based model? *Ecological Modelling, 272*, 141-149.

R Core Team (2019). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/

Schaub, M. & Kery, M. (2011). *Bayesian Population Analysis Using WinBUGS: A Hierarchical Perspective*. United Kingdom: Elsevier Science.

Skrade, P. D. B. & Dinsmore, S. J. (2010). Sex-related dispersal in the mountain plover (*Charadrius montanus*). *The Auk, 127*(3), 671-677.

Smith, P.A., Bart, J., Lanctot, R.B., McCaffery, B.J. & Brown, S. (2009). Probability of detection of nests and implications for survey design. *Condor, 111*, 414-423.

Spiegelhalter, D.J., Best, N.G., Carlin, B.P. & van der Linde, A. (2002). Bayesian measures of model complexity and fit. *Journal of the Royal Statistical Society: Series B (Statistical Methodology), 64*(4), 583-639.

Stroud, D.A., Baker, A., Blanco, D.E., Davidson, N.C., Delany, S., Ganter, B., Gill, R., González, P., Haanstra, L., Morrison, R.I.G., Piersma, T., Scott, D.A., Thorup, O., West, R., Wilson, J. & Zöckler, C. (2006). The conservation and population status of the world's waders at the turn of the millennium. In: *Waterbirds around the world*, The Stationery Office, Edinburgh, UK, 643-648.

Tipton, H.C., Doherty, P.F. & Dreitz, V.J. (2009). Abundance and density of mountain plover (*Charadrius montanus*) and burrowing owl (*Athene cunicularia*) in eastern Colorado. *The Auk 126*, 493-499.

Tipton, H.C., Dreitz, V.J., & Doherty, P.F. (2008). Occupancy of mountain plover and burrowing owl in Colorado. *Journal of Wildlife Management, 72*(4), 1001-1006.

U.S. Fish and Wildlife Service. (1999). Proposed Threatened Status for the Mountain Plover. Federal Register 64(30), 7587-7601, 16 February 1999.

Wunder, M.B., Knopf, F.L. & Pague, C.A. (2003). The high-elevation population of mountain plovers in Colorado. *The Condor 105*, 654-662.

Youngflesh, C. (2018). MCMCvis: Tools to visualize, manipulate, and summarize MCMC output. *Journal of Open Source Software, 3*(24), 640.

CHAPTER 4.

Problems Surrounding Development of Residential Structures on Wetland Habitat and Recommendations for Improvement, City of Newark, CA

Wetlands not only provide habitat for wildlife, but also protect human health by filtering water and shielding against flood waters (Keddy, 2010). The natural flood protection wetlands provide is particularly important for areas vulnerable to rising sea levels, such as coastal regions (Runting et al., 2016). Areas containing wetland habitat are commonly targeted for land development activities, and the proponents who stand to benefit from these actions tend to view development over these areas as inconsequential, not fully comprehending the repercussions that result from the loss of those vital services wetlands provide (Hu, et al., 2017).

In order to support a growing urban population, the City of Newark located adjacent to the San Francisco Bay in California has decided to build residential structures over floodplain wetlands found along an already highly developed and segmented estuary (Figure 3). Developing near and/or within floodplain wetlands can detrimentally affect the ability of the wetlands to filter water, to protect against rising sea levels and to provide habitat for endangered species, services that humans such as land developers take for granted (Keddy, 2010). Moreover, wetland loss also negatively affects organisms that use floodplain wetlands as habitat (Opperman et al., 2010). Although opposition against development near or within wetlands grows, particularly in the City of Newark case, the demand for space for development is considered to be of equal importance by many stakeholders. The City of Newark residential development project proposes that contractors move forward with construction near vital floodplain wetland habitat, therefore I propose that the current project plan goes through additional revisions. These revisions should focus on minimizing the amount of structures built on these floodplain wetlands by moving proposed developments away from wetland boundaries and to areas that possess low wetland cover. Doing so will minimize the impact of these developments on the vital functions these wetlands provide. In addition, supplementary environmental impact reports that incorporate proven wetland protection guidelines will be necessary to ensure the long-term sustainability of these wetland habitats.

For decades wetlands have been converted to suitable land for human development. Whether it be for agricultural, residential or commercial purposes, the result is the same, complete destruction of wetland habitat and the biota residing there (King, 1998). As the human population exponentially grows, more of the global population begins to reside in urban areas, thereby increasing demand for space to house new city inhabitants. When cities grow, urban infrastructure (e.g. bridges, culverts, roads and pipes) is frequently integrated with large-scale natural features, such as rivers, mountains and lakes (Wikum & Shanholtzer, 1978). Because such large-scale features dominate the landscape, they are less likely to be targets for development compared to small-scale natural resources, such as ponds, tree stand patches, headwater streams and wetlands, which are more easily razed (Wikum & Shanholtzer, 1978). Wetlands in particular have taken the brunt of these societal endeavors; today more than half of the world's original wetlands no longer exist (Dahl, 2011). As the City of Newark and proprietors continue to pursue their residential development objectives based on the current proposal, the inevitable loss of valuable floodplain wetlands appears to be an inescapable reality.

The City of Newark proposed to build this residential development because it projected an increase of 2.3 million people between 2010 and 2040. Almost one-quarter of this projected growth was realized by 2015 and has continued unabated since that time (Kroll et al., 2017). The homes in this proposed development will supply some of these new residents with housing. The current proposal (Newark Specific Plan, 2009) consists of 469 single-family homes, three park parcels and four boardwalks that overlook a 430-acre site known as Area 4 (Figure 1 & 2), located near the southwestern border of the city. Currently, Area 4 is predominantly undeveloped comprising seasonal wetlands and marshes. The city's general plan allows for the development of four "villages" that will share up to 874 units of low-density housing, 2,739 parking spaces, an 18-hole golf course and a park on the site.

The residential development project has been under consideration for development since the 1980s (Newark Specific Plan, 2009). In 1999, when development of the parcel was imminent, a proposed ballot measure to protect this open space failed, but the city's general plan was revised to allow development north of Area 4 instead. In 2015 the environmental impact report indicated no significant impact to the environmental resources in Area 4 (City of Newark, 2015), the Newark City Council approved plans for its development. However, a local citizens group, the Citizens Committee to Complete the Refuge, sued to stop the project on the grounds that the environmental impact report inadequately assessed the impacts (Simons, 2019). After losing the lawsuit, developers redrew the plan to create tiny islands of homes surrounded by flood-prone wetlands in an attempt to skirt regional regulations and state guidance that limit development in floodplains (California Coastal Act, 1976). The Newark City Council approved this plan on November 19th, 2019.

The City of Newark approved this plan because it created a revenue opportunity by selling off the city-owned land to meet the increased demand for affordable housing throughout the Bay Area. The East Bay cities of Newark and Dublin are among the 10 fastest-growing cities in San Francisco Bay with populations over 30,000 (Kroll et al., 2017). These cities boast locations near Silicon Valley, low crime and cheaper housing, making them convenient for local businesses and Silicon Valley commuters. A one-bedroom apartment in Newark rents for about $2,400 per month, while a single-family home sells for $750,000 on average (Kroll et al., 2017), a relatively low estimate compared to Newark's counterpart Silicon Valley. Consequently, The Sobrato Organization, the firm tasked with developing Area 4 stands to profit significantly from this lucrative real estate venture. New housing developments benefit local economies by increasing property values, bringing new critical infrastructure, improving the affordability of housing, and increasing revenue for local shops and services (Whitehead et al., 2015). Silicon Valley has extremely high living costs and potential employees for the companies that encompass that region may be unable to afford living costs in such an affluent area. The increase in affordable housing made available by developing Area 4 in Sanctuary West will accommodate the employees of several Silicon Valley companies. With the approval of this development, The Sobrato Organization, Silicon Valley employees, outside consultants including the survey crew to delineate wetlands, and the city government of Newark stand to benefit from this project, whether it be by direct revenue or a place to live. Although many stand to benefit from this development, the subsequent repercussions will come at great cost to the area's wetlands.

Stakeholders who stand to benefit from the project should also recognize that wetlands provide important ecological functions which underly ecosystem services of direct value to residents. These include providing habitat for a wide variety of flora and fauna; filtering, cleaning and storing water; collecting and holding flood waters; sequestering atmospheric carbon; and providing places of beauty and recreation (Cronk & Mitsch, 1992). Following European settlement, approximately 46% of wetland systems were destroyed by humans in the

U.S. (Dahl, 2011). The extent of this devastation is alarming: nearly 87% of the entire planet's original wetland habitats have been eradicated in the past 300 years, with approximately 54% of wetlands lost in the last 30 years (Hu et.al., 2017). California has destroyed more than 90% of the estimated 4 million acres of wetlands that once spread across the state by damming rivers and converting floodplains into farms and cities (Brown & Pasternack, 2005). In recent decades, humans have drained and/or filled wetlands to provide land for the exponentially growing human population, who utilized it for agriculture, residential development, mining, and dumping (King, 1998). Residential development of the floodplain wetlands, such as those in Area 4, may decrease water quality, destroy habitat for wetland specialists, disconnect these wetlands from the floodplains, favor excessive aquatic plant growth, or enhance establishment of invasive species (Kneitel & Lessin, 2009). Given the loss of such critical ecosystem services that benefit human well-being, many groups such as scientists and wildlife managers have opposed this proposal, including a prominent environmental group, the Greenbelt Alliance.

In order to construct the 469-residential structures, contractors would have to pave over a rare habitats on the South San Francisco Bay shoreline that support an abundance of wildlife including several endangered species. Because they prize the unique diversity of this area, scientists and wildlife managers have called for the inclusion and restoration of the Mowry Slough and adjacent uplands in the Don Edwards San Francisco Bay National Wildlife Refuge (USFWS, 2013). Over a dozen environmental organizations formally oppose the City of Newark's development proposal (Foote, 2019), because of the dangers the project poses to the threatened and endangered wildlife inhabiting the area. By approving a sprawling development in these floodplain wetlands, the Newark City Council will destroy vital habitat that support threatened and endangered species, including the salt marsh harvest mouse, California

Ridgway's rail, California black rail, and burrowing owl (USFWS IPaC), resulting in the loss of biodiversity.

The project area already sits in a high-risk area threatened by sea-level rise; moreover, the proposed development lies in a FEMA-designated flood zone, where new homes will be at a significantly increased risk of flood damage from the moment they are built (Newark Specific Plan, 2009). The cost to protect residents from flooding if these wetlands are destroyed will be substantial compared to the free flood protection the wetlands currently provide. Such costs would strain the budgets of both the City and its residents. Currently, Area 4 is only protected from San Francisco Bay water inundation by pumps and a network of public and private levees. 1.6 million cubic yards of fill, an estimated 100,000 truckloads, will need to be imported to lift homes out of the flood zone by 15 feet, the minimum mandated by the City council's specific plan (Newark Specific Plan, 2009). The Greenbelt Alliance group argues that officials need to focus on protecting natural infrastructure, like wetlands, that can buffer surrounding communities from rising sea levels, rather than spending millions of taxpayer dollars to construct residences in a highly flood prone area.

Scientists, conservationists and informed environmental activist groups like the Greenbelt Alliance have also argued that this project not only disturbs natural wetlands, but that these homes will sit on lands vulnerable to liquefaction. This highly saturated area requires draining every year to avoid flooding, which can be especially problematic during seismic events in an already earthquake prone region. With so much development currently occurring in the Bay Area, this new project is perceived as unnecessarily damaging by most groups opposing the proposal. Groups such as Save The Bay and Citizens Committee to Complete the Refuge, have raised alarms about these facts on numerous occasions, citing the project as costly for all when analyzing both the environmental and economic costs and benefits in their entirety (Foote, 2019).

Despite these environmental concerns, the City of Newark's 2015 environmental impact report (City of Newark, 2015) for the project contends that the surrounding floodplain wetlands will be unaffected by the project. Thus, no mitigation plan has been proposed or mentioned in any of the public documents. However, an examination of the project map (Figure 1 & 2) reveals that the boundary of the development clearly overlaps wetland boundaries, confirming that the finding of "no significant effect" in the report is biased as many groups have argued. Even if mitigation had been proposed, it would likely not adequately compensate for the lost natural wetlands in this project.

By law, land developers are required to mitigate for any net wetland loss in the United States (Section 404 CWA, 1972), but constructed wetlands do not provide the same services as natural wetlands. If the City of Newark were to provide plans to mitigate for wetland loss, wetlands created for mitigation will not provide the same degree of protection as the natural floodplain wetlands slated for destruction by this proposal (Moreno-Mateos, et al., 2012, Grossmann, 2012, Cole & Brooks, 2000). This notion was substantiated by hydrologic data collected through a long-term monitoring program in Florida that concluded that artificially created wetlands do not support similar levels of biodiversity or water filtration as natural wetlands do (Moreno-Mateos, et al., 2012). Furthermore, the designated areas where these mitigated wetlands are typically constructed include urban areas near parking lots or on roadsides, which at best serve as polluted reservoirs (Davis et al., 2010). For these reasons, the efficiency and legitimacy of the wetland mitigation process has been hotly debated (Hossler et al., 2011, Zedler, 1996). Natural floodplain wetlands are necessary for the protection of the City of Newark and the Bay area.

A clear recommendation for the project must be made based on the competing needs for housing and wetland protection. By analyzing this situation from a purely economic perspective, this development is very promising for potential Silicon Valley employees and for the general population based on housing needs. Despite these economic benefits, the estimated economic losses from sea level rise is quite large because the water storage and filtering functions of these wetlands will be lost. By viewing this situation solely from the environmental viewpoint, this project seems irreversibly damaging. As climate warms, sea-level rise in the Bay Area becomes more likely. If the proposed development destroys these wetlands, this project will inevitably degrade the protection the wetlands provide to coastal regions. Because current projections estimate sea-level will rise ~15 cm within the next decade this housing development is at risk of inundation (Griggs et al., 2017), making the long-term viability of the project improbable. If natural wetland functions are lost, water quality will be significantly degraded. Moreover, the newly built developments will potentially have a shortened lifespan based on sea-level rise.

Based on these facts, I recommend that the proposal be revised. First, the current proposal boundary for the development should be moved further inland to the north-northeast, thereby removing the floodplain wetlands out of the way of construction activities. By moving the boundaries to avoid impacts to wetlands, the total size of the development would be reduced, and a more suitable parcel should be found. One such parcel is located southeast of Area 4. This parcel is an optimal choice because it is currently not in use, meets the same zoning requirement, contains no threatened environmental resources and has the same value as Area 4 (Alameda County Assessor’s Office, 2020). Finally, following any changes to the project plans, a new environmental impact report should be written to include proven guidelines for development in close proximity to wetlands, such as those of the American Planning Association Policy Guide on Wetlands (APA, 2002). Guidelines from the APA that must be considered in the new reports include: promoting the inclusion of wetlands in the overall planning, recognizing the hierarchy of wetland protection techniques, and prioritizing minimization of wetland impacts over mitigation (APA, 2002). Because the new location is similarly priced to Area 4, my recommendation would only require revision of the environmental impact report. This is a small price to pay given the immense benefit afforded by ensuring the wetlands are protected. The bottom line is that plans for the project must change in order to protect these valuable floodplain wetlands and the l valuable ecosystem services they provide.

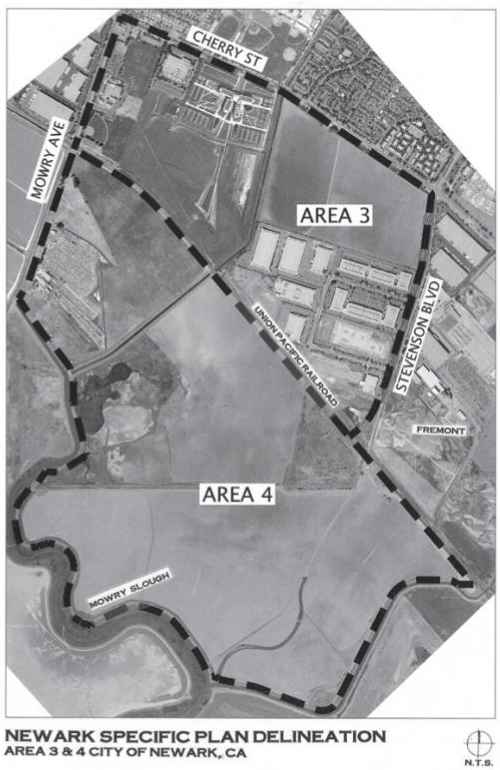

Figure 1. Area 4 of the proposal for residential developments

Figure 2. Newark's Specific Plan Land Use Diagram, depicting the boundaries of development and structures placement

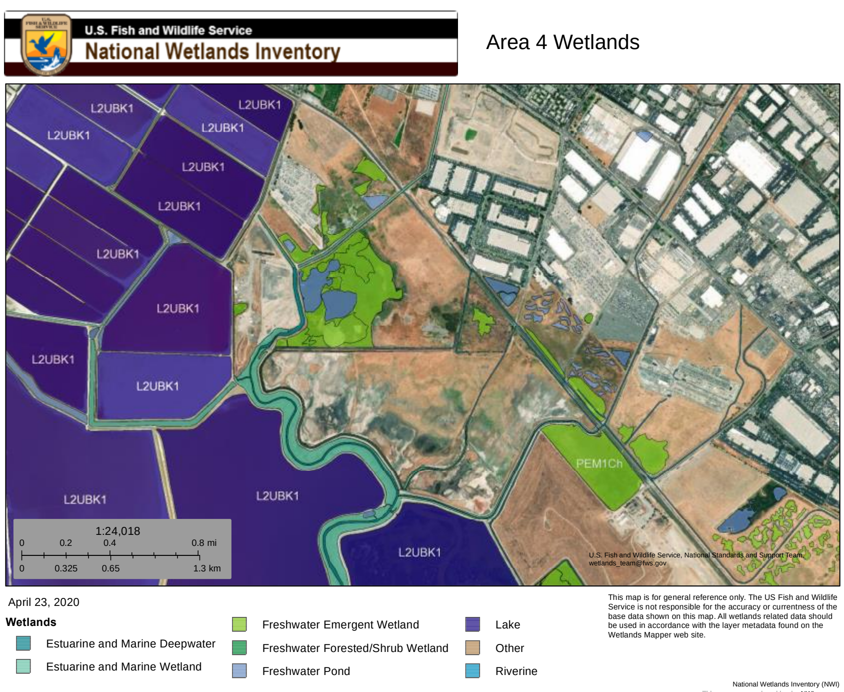

Figure 3. National Wetland Inventory (NWI) map depicting wetland habitat within Area 4 and the surrounding bay/estuary region

References

American Planning Association (APA). (2002). Policy Guide on Wetlands. https://www.planning.org/policy/guides/adopted/wetlands.htm.

Alameda County Assessor's Office. (2020). Property Assessment Information. Retrieved from https://acgov.org/MS/prop/index.aspx.

Arulrajah, A., Yaghoubi, E., Imteaz, M. & Horpibulsuk, S. (2017). Recycled waste foundry sand as a sustainable subgrade fill and pipe-bedding construction material: engineering and environmental evaluation. *Sustainable Cities and Society, 28*, 343-349.

Brown, K.J. & Pasternack, G.B. (2005). A palaeoenvironmental reconstruction to aid in the restoration of floodplain and wetland habitat on an upper deltaic plain, California, USA. *Environmental Conservation, 32*(02), 103-116.

California Coastal Act (CCA). (1976). California Public Resources Code, 30000 et seq.

Calil, J., Beck, M.W., Gleason, M., Merrifield, M., Klausmeyer, K. & Newkirk, S. (2015). Aligning natural resource conservation and flood hazard mitigation in California. *PLOS ONE, 10*(7), e0132651.

City of Newark. (2015). *Recirculated Final Environmental Impact Report*. https://www.newark.org/home/showdocument?id=182.

Cole, C.A. & Brooks, R.P. (2000). A comparison of the hydrologic characteristics of natural and created mainstem floodplain wetlands in Pennsylvania. *Ecological Engineering, 14*(3), 221-231.

Collinge, S.K., Ray, C. & Gerhardt, F. (2011). Long-term dynamics of biotic and abiotic resistance to exotic species invasion in restored vernal pool plant communities. *Ecological Applications, 21*(6), 2105-2118.

Cronk, J.K. & Mitsch, W.J. (1992). Creation and restoration of wetlands: some design consideration for ecological engineering. *Soil Restoration*, *17*, 217-259.

Dahl, T. E. (2011). Status and trends of wetlands in the conterminous United States 2004 to 2009 [PDF file]. U.S. Department of the Interior, Fish and Wildlife Service. Retrieved from https://www.fws.gov/wetlands/Documents/Status-and-Trends-of-Wetlands-in-the-Conterminous-United-States-2004-to-2009.pdf. (Original work published 2011).

Davis, A.Y., Pijanowski, B.C., Robinson, K. & Engel, B. (2010). The environmental and economic costs of sprawling parking lots in the United States. *Land Use Policy, 27*(2), 255-261.

Erwin, K.L. (2008). Wetlands and global climate change: the role of wetland restoration in a changing world. *Wetlands Ecology and Management, 17*(1), 71-84.

Foote, J. (2019). Newark luxury development would pave over restorable wetlands. Sustainable Communities, *Sierra Club San Francisco Bay*. https://www.sierraclub.org/san-francisco-bay/blog/2019/11/newark-luxury-development-would-pave-over-restorable-wetlands.

Gillespie, J. (2015). Catch 22: wetlands protection and fishing for survival. *Geographical Research, 54*(3), 336-347.

Goals Project. (1999). Baylands Ecosystem Habitat Goals. A report of habitat recommendations prepared by the San Francisco Bay Area Ecosystem Goal Project. First Reprint. San Francisco, CA/Oakland, CA: U.S. Environmental Protection Agency/San Francisco Bay Regional Water Quality Control Board.

Griggs, G., Árvai, J., Cayan, D., DeConto, R., Fox, J., Fricker, H.A., Kopp, R.E., Tebaldi, C. & Whiteman, E.A. (2017). Rising seas in California: an update on sea-level rise science. California Ocean Science Trust.

Grossmann, M. (2012). Economic value of the nutrient retention function of restored floodplain wetlands in the Elbe River basin. *Ecological Economics*, *83*, 108-117.

Hossler, K., Bouchard, V., Fennessy, M. S., Frey, S.D., Anemaet, E. & Herbert, E. (2011). No-net-loss not met for nutrient function in freshwater marshes: recommendations for wetland mitigation policies. *Ecosphere, 2*(7), 1-36.

Hu, S., Niu, Z., Chen, Y., Li, L. & Zhang, H. (2017). Global wetlands: Potential distribution, wetland loss, and status. *Science of The Total Environment, 586*, 319-327.

Junk, W.J., An, S., Finlayson, C.M., Gopal, B., Květ, J., Mitchell, S.A., … Robarts, R.D. (2012). Current state of knowledge regarding the world's wetlands and their future under global climate change: a synthesis. *Aquatic Sciences, 75*(1), 151-167.

Keddy P.A. (2010). *Wetland Ecology: Principles and Conservation*. New York: Cambridge University Press.

Khatri, T.B. (2014). Wetlands, biodiversity and climate change. *The Initiation*, *5*(0), 138-142.

King, J.L. (1998). Loss of diversity as a consequence of habitat destruction in California vernal pools. *California Native Plant Society*. Proceedings from a 1996 Conference, Sacramento, CA.

Kneitel J.M. & Lessin C.L. (2009). Ecosystem-phase interactions: aquatic eutrophication decreases terrestrial plant diversity in California vernal pools. *Oecologia, 163*(2), 461-9.

Kroll, C., Jaramillo, J., Lu S., Olsen, A. & Wong, H. (2017). Plan Bay area 2040: final regional forecast of jobs, population and housing. Metropolitan Transportation Commission; Association of Bay Area Governments

Mitsch, W.J., Bernal, B., Nahlik, A.M., Mander, Ü., Zhang, L., Anderson, C.J., … Brix, H. (2012). Wetlands, carbon, and climate change. *Landscape Ecology, 28*(4), 583-597.

Moreno-Mateos, D., Power, M.E., Comín, F.A. & Yockteng, R. (2012). Structural and functional loss in restored wetland ecosystems. *PLoS Biology, 10*(1), e1001247.

Newark Specific Plan. (2009). Appendix A, Newark Areas 3 and 4 Specific Plan. Area 3 & 4 of the General Plan.

Opperman, J.J., Luster, R., McKenney, B.A., Roberts, M. & Meadows, A.W. (2010). Ecologically functional floodplains: connectivity, flow Regime, and scale. *JAWRA Journal of the American Water Resources Association, 46*(2), 211-226.

Runting, R.K., Lovelock, C.E., Beyer, H.L. & Rhodes, J.R. (2016). Costs and opportunities for preserving coastal wetlands under sea level rise. *Conservation Letters, 10*(1), 49-57.

Section 404 of the Clean Water Act, 33 U.S.C. 1344. (1972). https://www.epa.gov/cwa-404/clean-water-laws-regulations-and-executive-orders-related-section-404

Simons, E. (2019). As the sea rises and climate changes, a Bay Area City approves 469 single family homes on a Bayshore flood zone. Climate Change, *Bay Nature*. https://baynature.org/201 9/11/14/as-the-sea-rises-and-climate-changes-a-bay-area-city-plans-469-single-family-homes-on-a-bayshore-flood-zone/.

U.S. Fish and Wildlife Service. Information for Planning and Consultation (IPaC). ecos.fws.gov. https://ecos.fws.gov/ipac/.

U.S. Fish and Wildlife Service. National Wetland Inventory (NWI). fws.gov. https://www .fws.gov/wetlands/.

U.S. Fish and Wildlife Service. (2013). Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California. Sacramento, California.

Wikum, D.A. & Shanholtzer, G.F. (1978). Application of the Braun-Blanquet cover-abundance scale for vegetation analysis in land development studies. *Environmental Management, 2*(4), 323-329.

Whitehead C., Sagor, E., Edge, A. & Walker B. (2015). Understanding the local Impact of new residential development: a pilot study. London School of Economics & Political Science.

Zedler, J.B. (1996). Ecological issues in wetland mitigation: an introduction to the forum. *Ecological Applications*, *6*(1), 33-37.